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Sediment Removal as a Lake Restoration Technique

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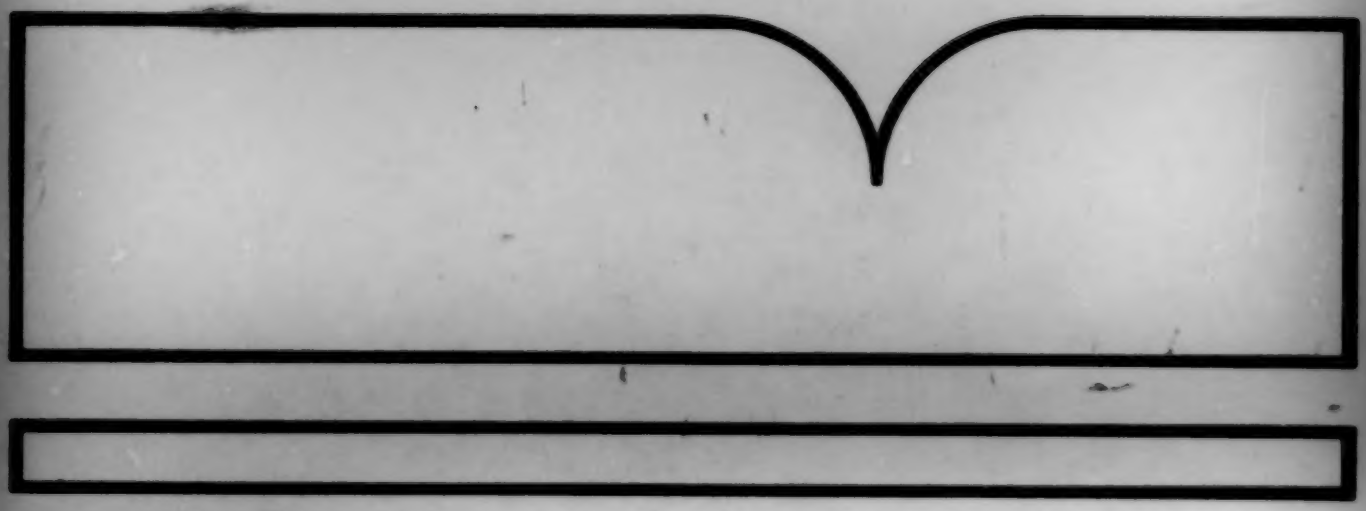

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SEDIMENT REMOVAL AS A LAKE RESTORATION TECHNIQUE

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ABSTRACT

Sediment removal as a lake restoration technique is reviewed to examine its positive and negative aspects. The effectiveness and longevity of the process is considered in terms of retarding eutrophication and reducing the impacts of toxic sediments. Freshwater lake sediment removal is usually undertaken to deepen a lake thereby increasing its volume to enhance fish production, to remove nutrient rich sediment, to remove toxic or hazardous material, or to reduce the abundance of rooted aquatic plants. Review of more than 60 projects and examination of five case histories reveals that the first three objectives are usually met through sediment removal. The effectiveness of dredging to control aquatic plants has not been well documented.

Advantages of sediment removal techniques include ability to selectively deepen part of a lake basin, increase the lake volume, recover organically rich sediment for soil enrichment, and improve limnetic lake quality. Disadvantages include high cost, phosphorus release from sediment, increased phytoplankton productivity, noise, lake drawdown, temporary reduction in benthic fish food organisms, the potential for release of toxic materials to the overlying water and for environmental degradation at the dredged material disposal site. High quality dredge material can be used for beneficial purposes and may offset the initial high cost of dredging.

The technique is recommended for deepening and for reducing phosphorus release from sediment. Sediment removal to control toxic materials is possible with minimal environmental impact when proper equipment is used, but it may be extremely expensive. Dredging will remove rooted aquatic plants, however, their re-encroachment rate will be depth, sediment texture, and sediment nutrient dependent. Lack of definitive information about the effect of these factors on the regrowth of various plant communities prohibits explicit recommendations on sediment removal to control rooted plants.

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I. Introduction

While control of external pollutant inputs to a lake should be given high priority in any restoration/protection plan, external controls alone may be insufficient to produce the desired changes in lake quality (Emery et al., 1973; Minder, 1948 in Peterson et al., 1976: Larsen et al., 1975). Where lake improvement goals cannot be met by watershed nonpoint source pollutant controls alone, sediment removal might be necessary.

Sediment removal usually includes one or more of four basic objectives. The first is simply to deepen a lake to improve boating, fishing, water skiing and other uses impaired by the effects of shoaling. A second objective is to prevent or reduce the internal cycling of nutrients. It has been shown that sediment recycling of nutrients in lakes may represent a significant fraction of the total nutrient loading to the lake (Funk, 1978; Larsen et al., 1975; Larsen et al., 1979; Cooke et al., 1977). A third objective might be to remove toxic sediment. A fourth objective, or possibly a fringe benefit of the first and second objectives, is to reduce nuisance aquatic macrophyte growths.

Until recently there was little documentation of the effectiveness of sediment removal for meeting these goals. Pierce (1970) reviewed 49 lake sediment removal projects and concluded that none provided sufficient data to determine the effects of lake dredging on the total lake environment. Four years later, Dunst et al. (1974) reported on 50 world-wide lake sediment removal projects, again pointing out a general lack of documented effects. Since 1974, additional lake sediment removal projects have been initiated. Some are complete with documented results while others are ongoing. The purpose of this paper is to review and update information on the effectiveness of sediment removal as a lake restoration technique. Personal communications are referenced liberally where documented results are unavailable and lakes of the United States are emphasized. The effectiveness of different types of sediment removal techniques and various objectives are illustrated by five case histories.

II. Purposes of Sediment Removal

1. Deepening

Sediment removal for maintaining navigation channels has been practiced for years. These deepening projects generally have been successful (Herbich, 1975). However, most have failed to address the source of the problem in the watershed, thus they necessitate rather frequent routine maintenance work. In small lakes, sediment removal may become necessary when uses such as boating are impaired as a direct result of shoaling. Other considerations pertaining to deepening are that lake volume must be sufficient to offset loss by seepage and evaporation. The U.S. Department of Agriculture (1971) recommends small lake and pond depths ranging from 1.5 m to 4.5 m for various parts of the country to compensate for water loss via these mechanisms. Toubier and Westmacott (1976) indicated that lakes in colder parts of the United States must be at least 4.5 m deep to avoid winter fish kills. Each of these considerations addresses sediment removal strictly from the standpoint of need to

deepen the lake basin. Pierce's (1970) survey of upper Midwestern dredging projects showed that projects undertaken for the purpose of deepening were all successful. The conclusion is not surprising, for who can argue that if sediment is removed from a lake basin, that the lake will be deeper?

2. Nutrient Removal

Nuisance algal growth and nutrient relationships in lakes are well documented; phosphorus is most frequently cited as the limiting nutrient (Bartsch, 1972; Porcella et al., 1974; Schindler, 1977). Uptake of nutrients from the water and sediment by plants ultimately results in deposition of organic materials into the lake basin. Table 1 shows the total phosphorus content of sediments in selected lakes in North America. A portion of this phosphorus may be released during spring and fall lake circulation in dimictic systems. In shallow polymictic lakes, sedimentary phosphorus release may be more frequent, constituting a greater nuisance by periodic infusion of nutrients to overlying water during summer recreation periods. The periodic influx of nutrients in this manner usually results in an over-abundant growth of unwanted phytoplankton causing reduced water transparency, undesirable green water color, and in severe cases a serious depletion of dissolved oxygen and fish kills. Livingston and Boykin (1962) estimated that sediment-regenerated phosphorus amounted to approximately 45% of the phosphorus loading to Linsley Pond, Connecticut. Welch et al. (1979) estimated the phosphorus loading from sediments in Long Lake, Washington to be 200-400 kg yr^{-1} or about 25%-50% of the external loading. Jones and Bowser (1978), citing a personal communication from Callender (1978), reported that sediment phosphorus regeneration in White Lake, Michigan constituted approximately 40% of that lake's phosphorus loading. Larsen and Schults (1978) stated that, "the supply of phosphorus from sources within a lake can be several times the supplies from external sources for periods of time on the order of months." Shagawa Lake, Minnesota experiences internally recycled summer phosphorus pulses of approximately 2000-3000 kg during June, July and August. This compares to an annual loading of phosphorus to the lake from wastewater (before advanced waste treatment) of 5000-5500 kg, and current loading of about 1000-1500 kg (Larsen, 1980). Thus, internal phosphorus loading to Shagawa Lake prior to advanced waste treatment (AWT) was about 28-35% of the total loading. Following implementation of AWT, internal phosphorus loading constituted approximately 66% of the total loading. Continued internal nutrient loading to the lake has been responsible for the apparent slow recovery rate.

It is evident that internal phosphorus loading to a lake can, in some instances, amount to major portions of the total loading. Thus, sediment removal to prevent internal loading should be weighed against alternative treatment techniques and given serious consideration where it might be expected to produce significant loading reductions.

3. Toxic Substances Removal

Sediment contaminated with toxic and hazardous materials poses a potentially severe problem for highly industrialized countries throughout the

Table 1. Total phosphorus content of sediments in selected lakes in North America (from Larsen and Schults, 1978)

Lake	Total Phosphorus (mg/g dry wt)	Core Depth	Reference
Sammamish, WA	2 - 5	40 cm core	Welch (1977)
Lower St. Regis, NY	0.5 - 1.4	35 cm core	Fuhs et al. (1977)
Huron	~1 - ~2	50 cm core	Kemp et al. (1972)
Ontario Great Lakes	~1.2 - ~3.0	50 cm core	Kemp et al. (1972)
Erie	~1 - ~2	50 cm core	Kemp et al. (1972)
	0.19 - 2.9	top 3 cm from 48 sites	Williams et al. (1976)
Monona, WI	~1 - ~2.2	100 cm core	Bortleson and Lee (1975)
Washington, WA	~1 - >6	40 cm core	Shapiro et al. (1971)
Shagawa, MN	1 - 5	160 cm core	Bradbury and Waddington (1973)

world -- "potential" because in many cases these problems simply have not been recognized. This picture is changing, however, as more thorough and sophisticated lake sediment surveys are conducted (Matsubara, 1979; Sakakibara and Hayashi, 1979; Horn and Hetling, 1978; Bremer, 1979; Mackenthun et al., 1979).

Theoretically, toxic materials in sediment might be controlled by covering or in-place detoxification. These treatment procedures are relatively untested. The obvious alternative is to remove the polluted sediment. The question is how to remove it without reintroducing toxicants into the water column and without causing secondary pollution problems at the disposal sites. Turner and Fairweather (1974) stated that "Once a body of water is polluted, the most effective means known to man to remove bottom sediment pollutants is an efficient, hydraulic dredge". Emphasis here should be placed on the word "efficient". While hydraulic dredges are the workhorse of the dredging industry, and are effective in moving large volumes of dredged material with minimal energy input, they are not always effective in minimizing the resuspension of bottom sediment. The key to removing polluted sediments is in minimizing sediment-water interface disruption, containment of polluted dredge materials

and treatment of decant water. Several mechanisms for attaining these goals have been described (Peterson and Randolph, 1977, 1978, 1979; Peterson, 1979; Barnard, 1978) and will be discussed later in this paper under sediment removal methods.

4. Rooted Macrophyte Removal

Rooted aquatic plants in the littoral zone of lakes often interfere directly with recreational uses. Fishing, boating and swimming become difficult or nearly impossible; further, an over-abundance of rooted plants may be aesthetically displeasing to some persons. Any one of these problems may be sufficient justification for removing part of the aquatic plant beds. Dredging is but one of several approaches, however, due to cost and convenience differences, harvesting and chemical treatments are more commonly used to "solve" these direct use impairment problems (Robson, 1974).

While direct use impairment problems may justify the removal of some macrophytes from lakes, there is an increasing literature on the effects of macrophytes on internal nutrient cycling. Their role in this process, with its effect on algal dynamics, may be an even more important reason for attempting to control macrophytes by removing them from a lake. Wetzel (1975) indicates that most of the organic matter found in small lakes may be derived from their littoral zone. Reimold (1972) showed that salt marsh plants (Spartina) act as nutrient pumps, extracting sediment nutrients, translocating them to stems and leaves and excreting them to the surrounding water on each successive tidal cycle. Other researchers have documented that various species of freshwater aquatic plants extract nutrients chiefly from the sediment and translocate them to the surrounding water (Schults and Malueg, 1971; Twilley et al., 1977; Carignan and Kalff, 1980).

There is now considerable evidence that healthy freshwater aquatic macrophytes do not excrete large quantities of nutrients to the surrounding water while in the active growth phase. They do, however, tend to concentrate sediment supplied nutrients in their tissues. Some loss is incurred through plant part sloughing, but major amounts of nutrients are recycled to the lake when plants fruit and during senescence and plant decay (Lie, 1979; Welch et al., 1979; Barko and Smart, 1980). Barko and Smart (1979) estimated that in-lake mobilization of phosphorus by Myriophyllum in Lake Wingra, Wisconsin might amount to approximately 60% of the annual external phosphorus loading. Welch et al. (1979) indicated that much of the "sediment" phosphorus loading in Long Lake, Washington was probably due to rapid plant die-off and decay.

While macrophyte removal may be incidental to the objectives of many lake deepening and nutrient control projects, it appears likely that the removal of these plants may be far more important to reducing in-lake nutrient cycling than had been suspected previously. Current evidence indicates that any long range lake restoration project concerned with in-lake nutrient controls will need to focus on both the macrophyte and sediment compartments (Carignan and Kalff, 1980; Barko, 1980).

III. Considerations for Sediment Removal

Currently there is no single, off the shelf, commonly accepted problem assessment procedure, remedial action plan, preferred piece of equipment or even agreement about when, for what reason, or how to remove sediment from a freshwater lake. Nevertheless, there are a number of factors which must be weighed when considering lake remedial action. Some are more critical to plans for sediment removal than they might be for other types of action. Factors which must be considered for any sediment removal project are: 1) determination of problem sources; 2) characterization of sediment; 3) determination of sediment removal depth; 4) environmental problems associated with sediment removal; 5) sediment removal methods; 6) sediment disposal area; and 7) lake conditions most suitable for sediment removal.

Clear-cut, definitive statements about each of the above factors cannot always be made. In some cases this may be due to basic lack of information. In others it will depend on individual lake circumstances such as the intended use of the lake, the geographic and socioeconomic setting of the area, and the project completion time-frame. The following discussion addresses each of the above factors.

1. Determination of Problem Sources

Lake problems generally are identified as a result of some use impairment. The use impairment problem may be identified by nothing more than the casual observation that the lake becomes excessively green with algae or choked with rooted aquatic plants at various times of the year. In the case of over-abundent algal growth the problem is most frequently traced to excessive amounts of phosphorus. Chapra and Tarapchak (1976) presented a cause and effect sequence of events related to phytoplankton problems in eutrophic lakes. The approximate sequence is that increased phosphorus loading produces increased mean annual phosphorus concentrations in water which lead to increased mean summer concentrations of chlorophyll a, which lead to increased water turbidity, increased Secchi depths, and finally to reduced dissolved oxygen levels when sedimented organic material decomposes. Macrophytes present a similar problem with regard to dissolved oxygen depletion and potential for fish kills. They differ, however, with respect to nutrient supply sources since most are derived from the sediment (Carignan and Kalff, 1980; Barko and Smart, 1980).

While casual observations of lake conditions may serve as the catalyst for action, they are insufficient for the development of lake restoration plans. More quantitative data are required to determine causes of perceived problems. One of the most common and useful approaches is a determination of nutrient budgets because algal and associated problems have been shown repeatedly to correlate with the nutrient level of lakes (Vollenweider, 1968; Vollenweider, 1976; Dillon, 1975; Larsen and Mercier, 1976; Dillon and Rigler, 1974). Reckhow (1979) critically reviewed a number of nutrient mass balance approaches and concluded that several will perform satisfactorily in predicting trophic state for a given set of lakes. However, Vollenweider's (1975) loading criteria have been most generally used due to ease of application in assessing the susceptibility of a lake to altered phosphorus loading.

Mass balances focus on methods to account for surface and subsurface nutrient inputs and outputs. They do not account directly for internal nutrient recycling, but can be used to estimate the relative importance of internal nutrient supplies. The internal gain or loss of nutrients can be calculated as the difference between the algebraic sum of the inputs and outputs of the lake and the change in nutrient concentration in the lake over the same time frame. Figure 1 illustrates the potential importance of internal nutrient cycling in shallow lakes and indicates how nutrient washout model predictions of lake recovery might vary from measured in-lake results.

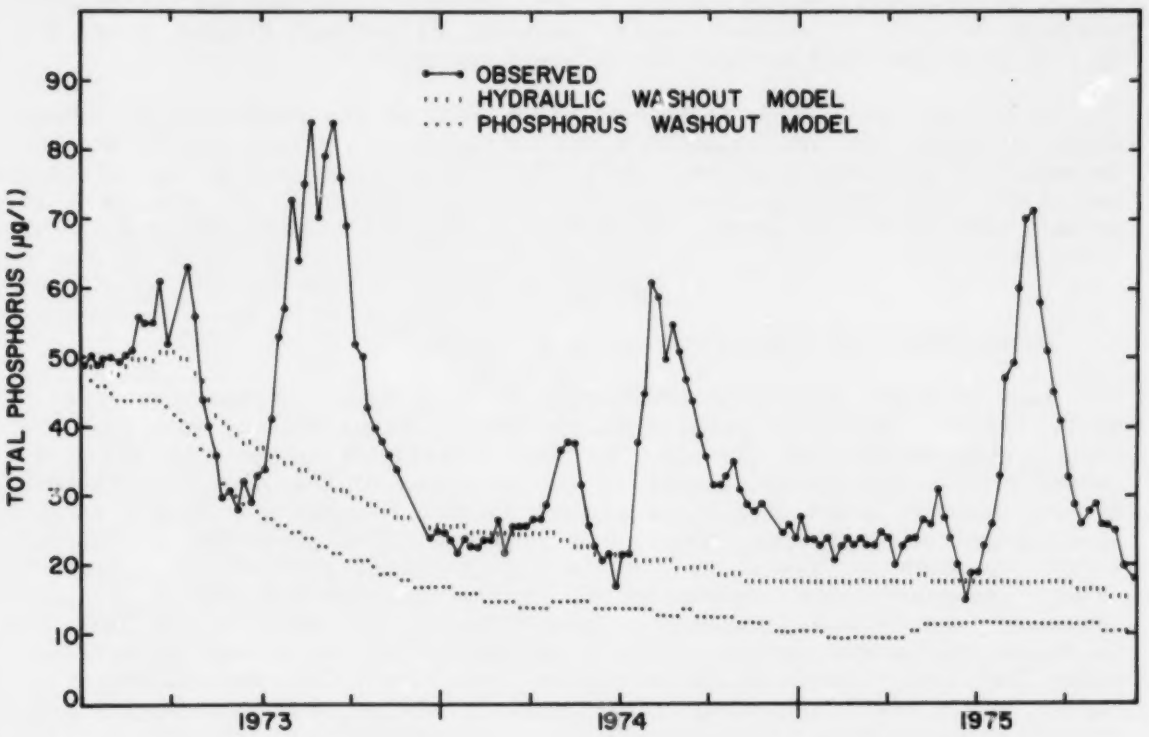


Figure 1. Comparison of hydraulic and phosphorus washout models with observations in Shagawa Lake, 1973-1974. Total phosphorus concentrations are the averages for the entire lake (updated from Larsen et al., 1975).

Larsen et al. (1979) used a modified form of the phosphorus residence time model to account for and accurately describe periodic summer pulses of internal phosphorus loading in Shagawa Lake, Minnesota. Application of modeling techniques or direct measures of nutrient release rates from sediment and calculation of the sediments' nutrient contribution to the lake is an important step toward determining the relative importance of the sediment compartment to the overall loading of the lake. Scheider et al. (1979) have summarized methods for using hydraulic and nutrient budgets to assess lake problems. A more recent and comprehensive review of the subject is presented by Reckow (1980).

2. Characterization of Sediment

If sediment removal appears to be a reasonable lake restoration option after an initial analysis, it may become necessary to further refine the sediment information base. It would be useful to determine various physical characteristics such as sediment depth and distribution, particle size and organic content, bulk density, and other factors critical to sediment removal. Chemically one might want to know if the sediment contains toxicants, what the nutrient content is and what the rate of release of nutrients from the sediment might be. A number of the physical and chemical variables can be determined on a one time analysis basis, however, where chemical release rates are involved the estimate must be made from more than one analysis. In the case of nutrients, these estimates may be made "directly" via time series analysis using two different approaches: 1) in situ isolation of portions of the lake bottom and overlying water suspected of contributing nutrients, or 2) incubation of sediment samples in the laboratory under simulated lake conditions (i.e., time, temperature, oxygen content, and pH). In either case, sediment nutrient release is measured as change in nutrient content of the overlying water with time. The following is a brief discussion of factors considered important to the description and analysis of phosphorus in sediments as a plant nutrient source. Much of the same rationale could be applied to an assessment of sedimented toxic materials.

Table 2 indicates considerable variation in anaerobic and aerobic phosphorus release rates for sediment from different lakes under in situ and laboratory conditions. Anaerobic and aerobic phosphorus release rates are both important since they represent conditions common to the two major sediment compartments of concern to the lake manager. These components are the deep water (profundal) sediments (Mortimer, 1941, 1942; Livingston and Boykin, 1962; Delfino et al., 1969) and the shallow water (littoral) sediments with associated plant communities (Lie, 1979; Cooke et al., 1977; Barko and Smart, 1979). The profundal sediments may alternate between anaerobic and aerobic conditions depending on lake stratification, while the littoral zone will usually remain aerobic.

Holdren and Armstrong (1980) evaluated the effects of mixing, redox, temperature, bioturbation, and sediment type on phosphorus release from profundal sediment. They found that bioturbation by tubificids and emerging chironomid larvae had the greatest effect on phosphorus release rates. This may be associated with Lee's (1970) observation that the sediment mixing zone extends 5 to 10 cm below the sediment-water interface in lakes with a well defined interface. In general, Holdren and Armstrong (1980) found that raising the

Table 2. Measurements of phosphorus release (modified from Holdren and Armstrong, 1980)

Sediment	Experimental System	Temp °C	Release Rate mg m^{-2} day^{-1} Aerobic	Anaerobic	Source[d]
Doboy Sound	Intact Core		0.031		Pomroy et al., 1965
Baldeggersee	In Situ		-9.3	9-10	Vollenweider, 1968
Clear Lake I	Laboratory[a]	27	9.7	8.7	Porcella et al., 1970
Clear Lake II	Laboratory[a]	27	7.4	7.3	Porcella et al., 1970
San Joaquin Delta	Laboratory[a]	27	13	11	Porcella et al., 1970
Lake Mendota	Laboratory[a]	27	49	53	Porcella et al., 1970
Lake Norriviken	Laboratory[a]		5	10	Ahlgren, 1972
Lake Trummen	Laboratory[a]			15[b]	Bjork, 1972
Lake of Tunis	Laboratory[a]			25-30[b]	Bjork, 1972
Lake Erie	In Situ	11	0.68	7.6	Burns and Ross, 1972
Simulated Benthal Deposit	Laboratory	22	3	154	Fillos and Molof, 1972
Ursee	Laboratory	10-15		6-16	Tessenow, 1972
Lake Ontario	Intact Core	4-10	0.03-0.8		Kemp-Nielsen, 1974
Fures	Intact Core	7	-2.0	17.3	Kemp-Nielsen, 1974
Lake Esrom	Intact Core	7	-1.4	12.3	Kemp-Nielsen, 1974
St. Gribso	Intact Core	7	0.2	1.2	Kemp-Nielsen, 1974
Grane Langso	Intact Core	7	0.6	0.8	Kemp-Nielsen, 1974
Lake Mendota	In Situ	7-13		7-11	Sonzogni, 1974
Buzzards Bay	In Situ	1.5	3.2		Rowe et al., 1975
Eel Pond	In Situ	20	4.9-16.0		Rowe et al., 1975
Lake Sodra Bargundasjon	Intact Core	8		36	Bengtsson, 1975
Muddy River	Laboratory	20-30	9.6	96	Fillos and Swanson, 1972
Lake Warner	Laboratory	20-30	1.2	26	Fillos and Swanson, 1972
White Lake	Intact Core			27.2	Freedman and Canale, 1977
White Lake	In Situ			19.0	Freedman and Canale, 1977
Lake Mendota	Intact Core	2-23	-1.9-33	0.67-65[c]	Holdren and Armstrong, 1980
Lake Wingra	Intact Core	4-21	-0.56-3.4	0.95-2.9[c]	Holdren and Armstrong, 1980
Lake Minocqua	Intact Core	3-18	<0.02-0.37	0.03-3.1[c]	Holdren and Armstrong, 1980
Little John Lake	Intact Cone	4-16	<0.02-1.1	0.02-3.8[c]	Holdren and Armstrong, 1980

[a] Columns containing dredged sediment and water.

[b] Not designated as either aerobic or anaerobic.

[c] N_2-treated cores with 2-3 mg l^{-1} dissolved oxygen in overlying water.

[d] Complete citations in Holdren and Armstrong, 1980.

temperature or lowering the oxygen concentration in overlying water also led to higher rates of nutrient release. Suspending the sediment or lowering the water temperature tended to remove phosphorus from the water column. They also reported that sediment type affected the relative importance of different incubation conditions. One example of this was that redox had more effect on noncalcareous sediments while temperature had more effect on calcareous ones. Most of Holdren and Armstrong's (1980) experiments ran for 10 days. Some were continued for 20-30 days. In most cases, phosphorus equilibrium was reached in 2 to 10 days. In some cases, the phosphorus levels increased slowly over the whole incubation time. They concluded that, given the proper environmental conditions, lake sediments can release significant amounts of phosphorus to overlying water, even when those waters are aerobic. Stauffer (1980a) provides detailed comments and interpretation of Holdren and Armstrong's results.

It is important to note that not all sediment phosphorus is exchangeable or biologically available. Some forms are tightly bound to sediment or actually incorporated as part of the structural matrix (Jones and Bowser, 1978). Therefore, total sediment phosphorus (bulk analysis) is of limited use in determining the potential for nutrient release. The phosphorus most likely to exchange readily with overlying water is that dissolved in the interstitial zone (pore water). Because dissolved interstitial phosphorus is often present at high concentrations (up to 5 mg/l in Shagawa Lake, Minnesota according to Larsen and Schults, 1978) in relation to the overlying water, there is a strong gradient for diffusion of the interstitial phosphorus to the overlying water. As this phosphorus moves to the water column it must be replaced by a repartitioning of particulate phosphorus. Some partitioning forms undergo transition more readily than others, thus they more readily replace liberated interstitial phosphorus and are therefore more important in terms of internal phosphorus recycling.

Williams et al. (1976) suggested that sediment phosphorus be divided into three components based on chemical extraction techniques (Figure 2): "apatite phosphorus" (A-P), "organic phosphorus" (O-P), and "nonapatite inorganic phosphorus" (NAI-P). The NAI-P is particularly important in lake sediment/water nutrient exchanges because it is thought to be strongly associated with iron as a ferric-oxide-orthophosphate complex under aerobic conditions. The iron complex dissociates under anaerobic conditions releasing phosphorus to interstitial water or directly to the overlying water. For this reason, Larsen and Schults (1978) have suggested that the amount of NAI-P in sediment might be a more valid measure of the readily available sediment phosphorus reservoir than either total phosphorus or pore water phosphorus. This is substantiated by Williams and Mayer's (1972) conclusion that an increase in apatite (phosphorus incorporated as part of the apatite crystalline structure) with depth in Lake Ontario sediments was a product of diagenesis of other forms of phosphorus to apatite with time. This transformation process and its relationship to more readily exchanged phosphorus forms is shown in Figure 3. Jones and Bowser (1978) indicate that processes of organic phosphorus mineralization and the release of sorbed phosphorus under reducing conditions in lake sediment can provide resolubilized phosphorus through diffusion from pore water, but that this process will be offset by apatite formation. The process may be responsible for the commonly observed phenomenon of decreasing quantities of soluble reactive phosphorus with increasing depth in lake sediments.

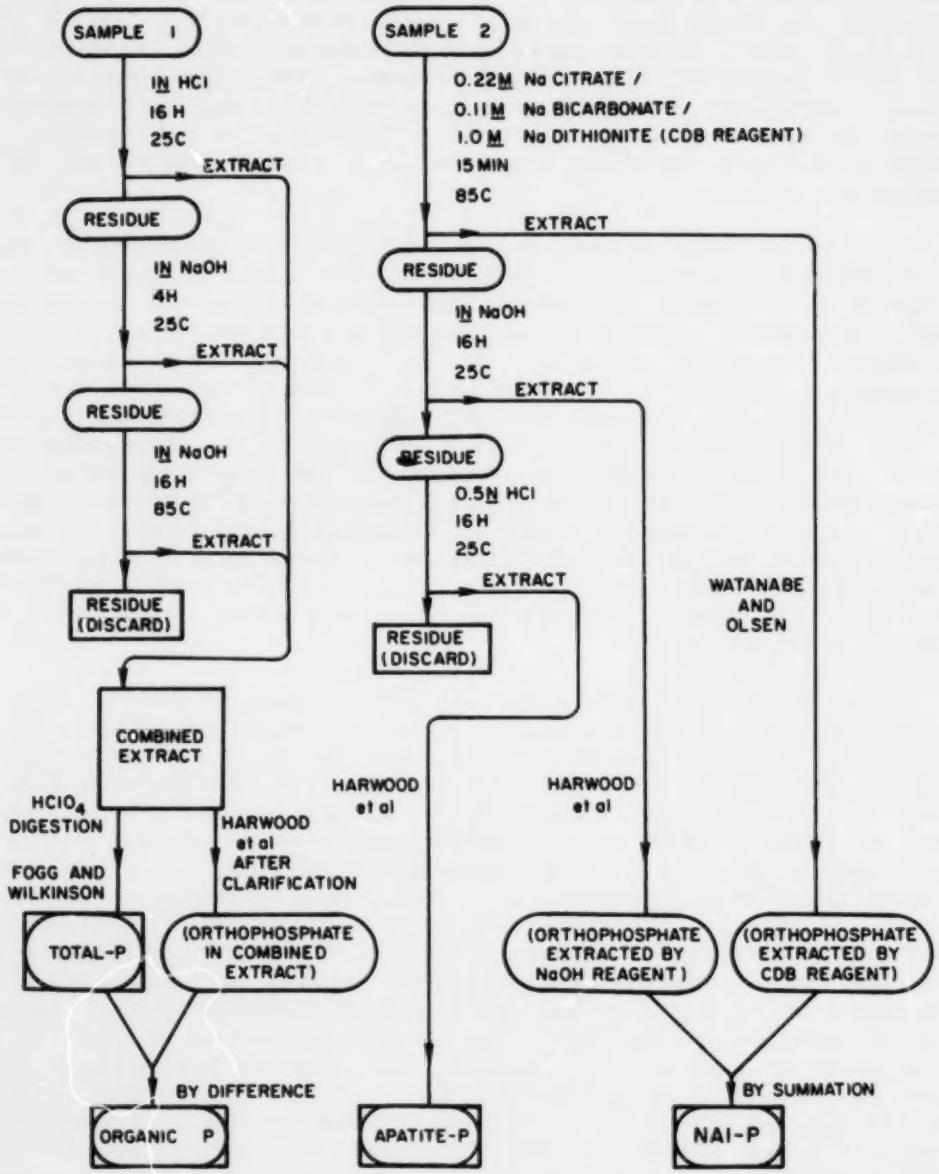


Figure 2. Outline of analytical procedures used to determine apatite-P, NAI-P, and organic P (modified from Williams et al., 1976).

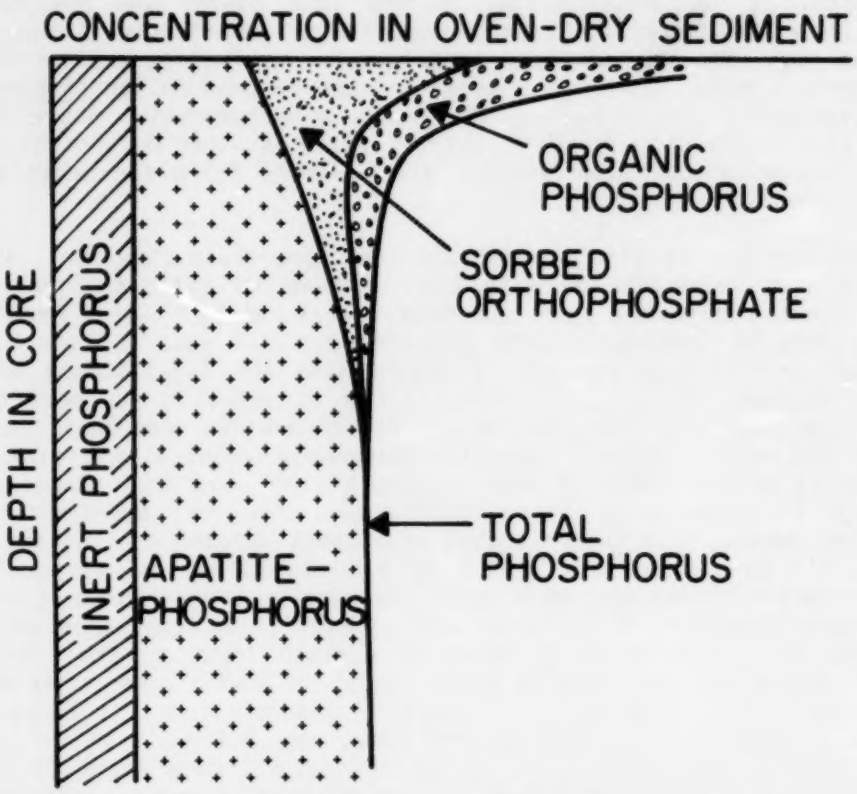


Figure 3. Distribution of phosphorus in a sediment profile from Lake Erie (from Williams and Mayer, 1972) hypothetically depicted and illustrating the diagenesis of mineral phosphate from amorphous to crystalline form. Percentage of each form is given by area fields (modified from Jones and Bowser, 1978).

One objective of sediment removal for nutrient control purposes might, therefore, be to skim the upper nutrient-rich layer of sediment off to reveal a strata containing a less readily exchangeable nutrient form. Characterization of pore water and solid phase sediment profiles from representative areas of the lake will permit the development of maps to locate the horizontal and vertical distribution of the relatively nutrient-poor strata. For phosphorus limitation planning these maps should be based on the component phosphorus contents of the sediments as proposed by Williams et al. (1976). Methods for measuring these components are described by Sommers et al. (1972) and by Williams and Mayer (1972). Interstitial water extraction methods are described by Holdren et al. (1977). Sediment characterization data coupled with time series temperature and oxygen profiles can be used to calculate the potential rate of nutrient release from sediments.

Determination of sedimentation rates of organic detritus and nutrient elements is important in providing estimates of how rapidly sediment has accumulated, or is accumulating, in the lake basin, and how rapidly the effects of sediment removal might be expected to deteriorate if the sedimentation rate remains high. A common technique for estimating sedimentation rates in the United States involves determining the ragweed pollen (Ambrosia) horizon in sediments and measuring the depth of accumulation above that known horizon (Bortleson and Lee, 1972; Craig, 1972). Another method for estimating sediment accumulation rates employs the isotopes ^{210}Pb (Koide et al., 1973) and ^{137}Cs (Pennington et al., 1973).

Jones and Bowser (1978) point out that pore fluid chemistry, solid phase chemistry and sedimentation rate may be used to calculate the fluxes of solutes (thus nutrients) into and from the sediments. Close examination of sediment data in conjunction with physical-chemical data for the lake should be useful to the lake manager in determining the significance of internal nutrient release and thus the advisability of removing sediment to control internal cycling. For example, if one has determined from the literature, in situ, or laboratory studies that the phosphorus release rate from anaerobic sediments is on the order of 5 to 10 mg m^{-2} day^{-1}, and that 50 percent of the lake basin in question is intermittently anaerobic for approximately 20 days during the summer, then the potential phosphorus loading to the lake from this source would be: 5 to 10 x 0.50 x 20 = 50 to 100 mg m^{-2} $summer^{-1}$. If the overlying water column was 10 m thick the resultant phosphorus loading to the water might amount to 5 to 10 mg m^{-3} $summer^{-1}$. A comparison of the above phosphorus release rates with those of surface input to the lake and with those of deeper sediment strata might assist in making a decision about sediment removal. For example, if it could be demonstrated that deeper sediment released phosphorus at a much slower rate, say 2.5 to 5 mg m^{-2} day^{-1}, the loading to the overlying water column from the strata might be reduced to approximately 2.5 to 5 mg m^{-3} $summer^{-1}$, or about a 50 percent reduction. This could translate to a similar magnitude reduction for chlorophyll a in the lake.

A similar approach could be used to assess the impact of vegetated and non-vegetated aerobic littoral sediments, taking into consideration the percentage of lake basin covered by each. In the case of vegetated areas (macrophytes) one should keep in mind that seasonal variation in nutrient release may be critical (Barko and Smart, 1980; Carignan and Kalff, 1980).

The above mentioned analytical approach provides a simplistic example of how a quantitative estimate of the effectiveness of sediment removal for nutrient control might be compared to other major loading sources to the lake. It also provides an idea of how long the effects (minus sedimentation rates) might be expected to last. Not all of the phosphorus released from the sediment is recycled to overlying water, so the potential impact is probably overestimated. Stauffer and Lee (1973), however, have shown that significant amounts of hypolimnetically released phosphorus do reach the epilimnion in shallow lakes as the thermocline is eroded by strong winds. One might improve the accuracy of estimates for sediment-released phosphorus impacts on the lake by applying Stauffer and Lee's mixing model approach, or by application of Stauffer's (1980b) "LAKETRANS" model which predicts thermal diffusivities and

solute transport in stratified lakes. A thorough discussion of internal nutrient loading in lakes and methods for assessing their significance was completed recently by Stauffer (1980a).

3. Determination of Sediment Removal Depth

When deepening a lake to improve it for sailing, power boating and associated activities, the deepening requirements are relatively straightforward. Projects aimed at controlling internal cycling of nutrients may focus on open water area sediments or on littoral sediments and associated macrophyte beds. Sediment removal requirements in these cases are not always clearly defined.

Lake Trummen, Sweden, is perhaps the most thoroughly documented case history of a lake subjected to sediment removal to control internal nutrient cycling. Sediment removal depth in Lake Trummen was determined in a fashion similar to the example given previously. Sediment was characterized chemically and physically, both horizontally and vertically, in the lake. Digerfeldt (1971) as referenced by Bjork (1972) determined that sedimentation rates in the lake increased dramatically during the period from 1940 to 1965. About 40 cm of loose FeS-colored (black) fine sediment was deposited during that time. Aerobic and anaerobic release rates of PO_4^--P and NH_4^+-N from the black sediment layer and the underlying brown layer were shown to be markedly different. Based on these differences a plan was developed to remove the upper 40 cm of sediment. Results of the project are presented as case history #1 in a later section of this paper.

Another approach to determining sediment removal depth has been used by Stefan and Hanson (1979) and by Stefan and Ford (1975). Their approach was to use a lake temperature model to predict the lake depth necessary to prevent summer destratification. The model uses air temperature, dew point temperature, wind direction, solar radiation, and wind speed inputs at 12-hour time step intervals, plus a consideration of lake morphology, to produce an output of temperature prediction with depth. The approach used by Stefan and Hanson (1979) is similar to that developed by Stauffer and Lee (1973) which shows how windpower erodes the thermocline of northern temperate lakes. The major difference is that Stefan and Hanson (1979) used their model output to predict to what depth a lake must be dredged to maintain a stable summer stratification, thus preventing enriched hypolimnetic waters from mixing into the epilimnetic zone.

Use of the Stefan and Hanson (1979) model for determining sediment removal depth assumes that a stable summer stratification is necessary to prevent enriched hypolimnetic waters from mixing into the epilimnion. Based on that assumption, Stefan and Hanson (1979) indicated that Hall Lake (one of the Fairmont, Minnesota lakes) would require dredging to a maximum depth of 8.0 m. Other data presented by Stefan and Hanson (1979) indicate that dredging to 8.0 m may require the removal of 4.0 to 6.0 m of sediment. There is no reference to the sediment volume involved. However, it would be sizable. Hall Lake has a surface area of 2.25 km^2 and a mean depth of 2.1 m. Lake profile data presented by Stefan and Hanson indicate an accumulation of 2.0 to 10.0 m of sediment in the lake basin overlain with about 0.5 m of "boundary

layer limnetic material". There is no apparent chemical or physical distinction between this layer and the deeper sediment. In fact, their core analyses show relatively uniform phosphorus concentration in sediment from the surface to a depth of 8.5 m (737 to 1412 mg kg^{-1} for 37 samples, with a mean of 1097). There is no indication, however, of phosphorus release rates for sediment from different depths. It is possible that the release rates from deeper sediment might be less than that of the surface sediment. Furthermore, it is possible that reduced nutrient release from the deeper sediment could be sufficient to significantly reduce the adverse impact of nutrients on the overlying water even though stratification was not stable (Bengtsson, et al., 1975). If that were the case, it argues for removal of much less sediment at significantly less cost and perhaps fewer disposal problems. Therefore, it seems advisable to examine phosphorus release rates for sediment from various depths before adopting a lake temperature modeling approach to determine dredging depth.

Where sediment removal is contemplated to control nuisance macrophyte growths and associated nutrient cycling, it is necessary to know what depth must be attained to prevent the nuisance. Guidelines for determining this variable are even less definitive than those for controlling nutrient cycling from limnetic zones.

A number of factors contribute to the growth of macrophytes, e.g., sediment type, nutrient availability, temperature and light level. The Wisconsin Department of Natural Resources presently uses light level (as determined by Secchi disc determination of water clarity) as the chief determinant of nuisance macrophyte growth. The method consists of determining the average summer water clarity through periodic Secchi disc measurements, and then estimating the maximum depth of macrophyte growth. The estimate is determined from a graphic display of maximum plant growth depth (m) as a function of Secchi disc (m). The straight line relationship is described by the equation:

$$Y = 0.83 + 1.22X$$

where, Y = maximum depth of plant growth in meters

X = average summer water clarity (Secchi disc) in meters.

The r^2 for this relationship is 0.53, with N equal to 55. According to Dunst (1980b) the relationship was derived from work done by Belonger (1969) and Modlen (1970). Dredging projects designed for macrophyte control in Wisconsin currently use this technique as a guide to prescribe dredging depths for macrophyte control (Dunst, 1980b). However, Dunst (1980a) also indicates that dredging depth need not always exceed the predicted Y-value to achieve effective rooted plant control. Simply increasing the water depth can result in speciation changes, and in Wisconsin, nuisance surface growths are uncommon where depth equals or exceeds three meters. Moreover, removal of soft organic sediment to hard sand and clay bottom usually results in reduced plant growth and density of problem species. Unfortunately, truly predictive methodologies are undeveloped in these areas. It should be pointed out that the objective of dredging for macrophyte control is not synonymous with macrophyte eradication. Any macrophyte dredging plan must take into consideration the preservation of fish spawning and nursery areas, waterfowl feeding areas, and other wildlife habitats.

4. Environmental Problems Associated with Sediment Removal

Peterson (1979) pointed out a number of potential environmental problems associated with sediment removal. The problems generally can be classified into in-lake and disposal site types. In-lake problems commonly center around the resuspension of sediment during its removal when dredges, drag lines, and clam shells are employed. One of the most common problems is the liberation of nutrients with resuspended fine sediment. Phosphorus is of particular concern due to its high concentration in interstitial waters in eutrophic lakes and its affinity for finely divided particulate material. Dredge agitation and wind action tend to move the disturbed nutrient-laden sediment into the euphotic zone of the lake, producing the potential for algal blooms. Churchill et al. (1975b) reported increased phosphorus concentration in Lake Herman, South Dakota, coincident with dredging but they noted no increased algal production. This was presumably due to the high turbidity level. Dunst (1980a); on the other hand, found increased algal production in Lilly Lake, Wisconsin when dredging began. This was a short term phenomenon associated with the immediate time of active dredging. There were no significant adverse effects.

Apparently, the reverse of increased algal production problems can also be triggered by the resuspension of sediment. Reduced light penetration due to turbidity will have a tendency to inhibit algal production. Lackey et al. (1959) indicated this problem may be further aggravated by the tendency of small phytoplankton to be adsorbed to the surface of precipitating fine sediment particles. A potentially more serious problem associated with finely divided sediment in the water column is oxygen depletion. If the sediment is highly organic, the particles quickly become bacteria-coated. The tremendous surface area of these particles permits rapid decomposition and possibly oxygen depletion. This factor may pose a problem in lake dredging projects considering that the organic content of lake sediment may approach 80 percent on a dry weight basis (Wetzel, 1975). To my knowledge there is no documentation of this problem in lake dredging projects, however, one should be aware of the potential for it.

Another problem associated directly with resuspended sediments is the liberation of toxic substances. Although small lake sediment removal projects for the specific purpose of removing toxic substances are relatively uncommon, the concern is becoming more evident (Matsubara, 1979; Sakakibara and Hayashi, 1979; Bremer, 1979). The major problem associated with toxicants and resuspended sediment centers around the fine particle sizes. Murakami and Takeishi (1977) have shown that when dredging in PCB (polychlorinated biphenyl) contaminated sediments, up to 99.7 percent of the PCB materials adhere to sediment particles less than 74 μ in diameter. This could be a particularly perplexing problem in freshwater dredging due to the increased particle settling times required for freshwater over those required for brackish or marine waters.

A relatively common concern with lake dredging projects involves destruction of the benthic community. If the lake basin is dredged completely, two to three years may be required to reestablish the benthic fauna (Carline and Brynildson, 1977). However, if portions of the bottom are left undredged the reestablishment may be almost immediate (Andersson et al., 1975) or within one

to two years (Crumpton and Wilbur, 1974). In any case, the effect on the benthic community appears to be of relatively short duration compared to the longer term benefits derived from sediment removal. The same appears to be true for game fish populations (Carline and Brynildson, 1977; Spitler, 1973).

The above concerns are associated primarily with dredging as a sediment removal technique. The other major technique for sediment removal in lakes is drawdown (lowering the water level) to expose the littoral sediments, or in some cases (Born et al., 1973) the entire lake basin, followed by removal of sediment with earth-moving equipment. Drawdown accompanied by bulldozer sediment removal may pose additional nuisance problems such as noise, dust, and truck traffic. It should be pointed out that hydraulic dredging might produce an unplanned lake drawdown if the solid to liquid ratio of the dredge slurry is low (typically 5% to 10%) and the dredge pumping rates are high.

5. Sediment Removal Methods

Once a lake has shoaled to the degree that it no longer serves a useful lake-oriented recreational purpose, the methods available for amelioration of the problem are limited. In theory, there are two approaches. One is construction of an outlet control structure to raise the water level. This approach has limited practical application due to problems of shoreline and habitat destruction and flooding of private property. The second and most common alternative is sediment removal.

The drawdown/excavate method of sediment removal was mentioned above. The technique is summarized by Born et al. (1973). It can be seen that the applicability of this method is limited to lakes with control structures or to those where pumping rates above water inflow rates can be achieved at reasonable cost.

Dredging is a much more common approach to sediment removal. Pierce (1970) described various types of hydraulic dredging equipment and provided guidance on the engineering aspects of dredge selection, i.e., size of dredge, pumping rate, pumping distances, etc. Barnard (1978) and Peterson (1979) have described various grab, bucket, and clam shell dredges; hydraulic cutterhead dredges; and specialized dredges to minimize secondary water quality impacts.

The conventional hydraulic cutterhead dredge (Turner and Fairweather, 1974) is the most commonly employed piece of dredging equipment. Hydraulic dredges have a number of advantages over grab or bucket type dredges. They have almost continuous operating cycles, allowing removal of large volumes of material in a short time. Production rates in excess of 1,000 m^{-3} hr^{-1} are not uncommon. These dredges can span virtually any horizontal reach of water, due to their floating operations platform and, in general, have the capability to be "environmentally cleaner" than grabs and buckets, due to their closed cycle system of operation. This may not always be the case, however, because by virtue of their ability to move large volumes of sediment, they also might create large turbidity plumes, with resuspension of sedimented nutrients and hazardous materials. The turbidity around the cutter of a hydraulic cutterhead dredge increases exponentially with the thickness of the cut, rate of swing, and cutter rotation rate (Barnard, 1978). The latter three functions

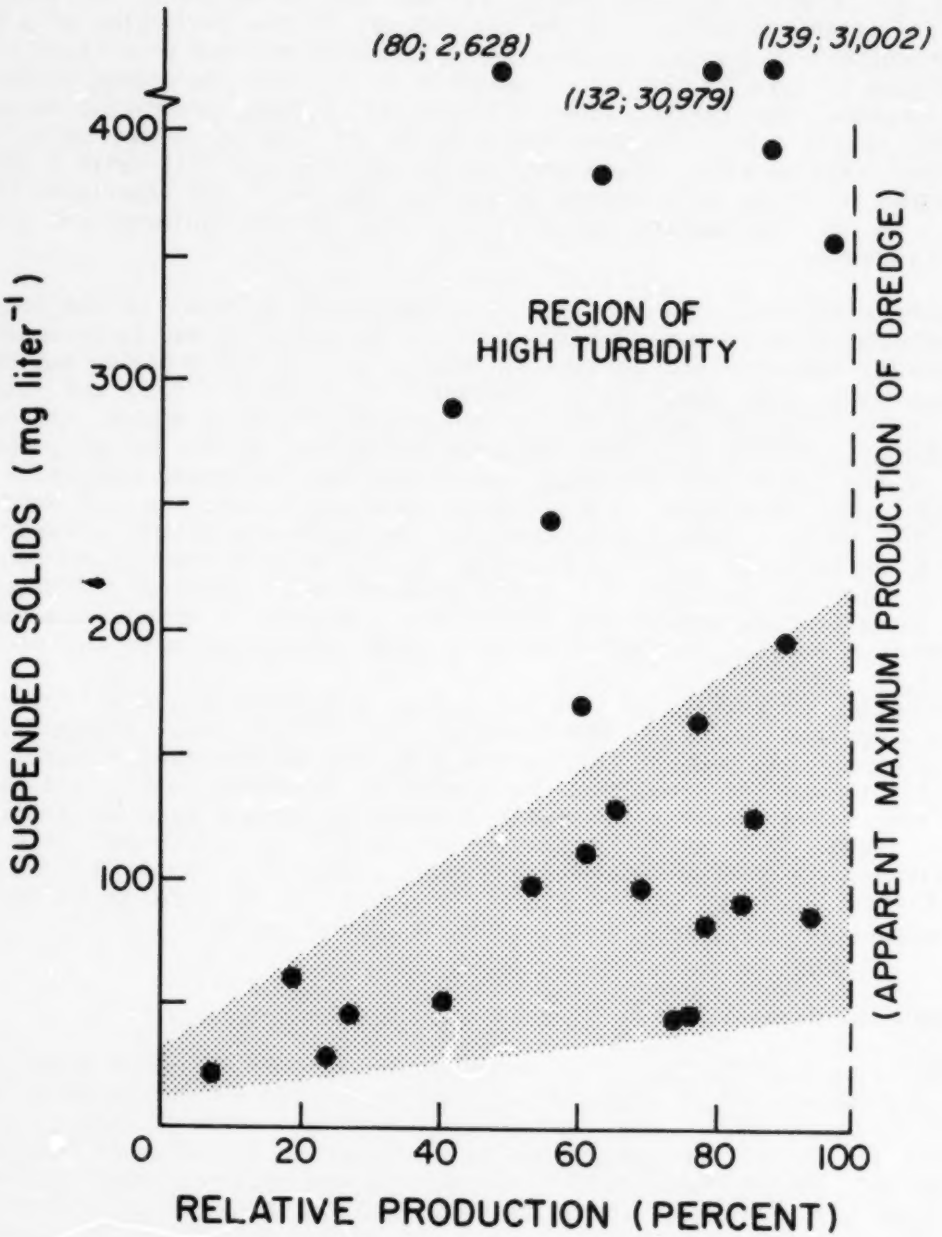


Figure 4. Relationship between the concentration of suspended solids 1 m from the cutterhead and the relative production of a 61 cm cutterhead dredge (from Barnard, 1978).

are related closely to dredge production rate. Therefore, production rate is closely related to turbidity around the dredge cutterhead. Using data from Yagi et al. (1975), Barnard (1978) plotted the relationship of suspended solids collected one meter from the cutterhead, to the production of a 61 cm diameter cutterhead dredge (relative to its apparent maximum production rate). He concluded (Figure 4) that it is possible to increase the dredge production to the maximum rate (broken line in Figure 4), without generating excessive turbidity levels. This involves optimization of cutting depth, swing rate, and cutter rotation rate. Therefore, minimization of turbidity with a cutterhead hydraulic dredge will depend as much on the skill and experience of the operator as on the texture and particle size of the sediment and general operating conditions.

Where minimizing resuspension of sediments is critical to the progress and completion of a project (toxic substances removal), it may be necessary to take special measures and perhaps to employ specialized dredging equipment. Equipment types and capabilities, described by Barnard (1978) and Peterson (1979), vary from modified grabs and innovative cutterhead design, the newest being the disc bottom cutterhead (Breebot brochure), to air driven pumps. A number of the specialized equipment variations have been used successfully in removing toxic substances from Japanese waterways (Peterson and Randolph, 1977, 1978, 1979). Suda (1979) compared the suspended solids concentration around a conventional cutterhead dredge to that around specialized Japanese dredges. Results indicated an approximate 10-fold reduction in suspended solids concentration around the specialized dredges. A serious drawback to these results, however, is that no comparisons of production rates were made.

Recently, scientists at the U.S. Army Engineers Waterways Experiment Station completed tests of the "pneuma" (air driven pump) dredge system. While no mention was made of this system's ability to minimize resuspension of sediments, it was indicated that the system was extremely inefficient because of its high energy input requirements relative to dredge material production (McNair, 1980). This may be indicative of the bid cost differentials between hydraulic and specialized dredging equipment reported by Peterson (1979). In view of this, one must examine closely how critical it might be to minimize the resuspension of sediments in any sediment removal project.

6. Sediment Disposal Area

The major non-lake impact of sediment removal centers around the disposal site. High population densities among a myriad of other things make it difficult to locate suitable sites. Another complicating factor is that the indiscriminate filling of low-lying areas is no longer permitted under Federal Law (Section 404, Public Law 92-500). Fill permits are required under Section 404 for virtually all projects where the lakes and any adjacent wetland proposed for filling exceeds 4.0 ha (10 acres) in area. Guidelines for preparing Section 404 permit applications are available from District Offices of the U.S. Army Corps of Engineers.

In upland disposal areas (where Federal permits are not required) diking operations are commonly employed. A major problem with upland disposal is dike failure (Calhoun, 1979). Another problem not uncommon to dredging projects is underdesign of disposal area capacity. This problem is not always

readily evident, but may become apparent as the project progresses. Disposal of freshwater sediments may be particularly troublesome because of their relatively slow settling rates (Wechler and Cogley, 1977). As disposal areas fill, their ponding depth is reduced and so is their efficiency to retain suspended solids. Disposal areas must be designed for the end of the project, not for some intermediate point. Palermo et al. (1978) have summarized important technical information and research projects which will assist with the proper design, construction, and maintenance of dredged material disposal areas. Where standards for discharge from disposal areas cannot be met, it may be necessary to treat the discharge. Barnard and Hand (1978) describe when and how to treat if it becomes necessary. Other reports prepared by the U.S. Army Corps of Engineers which are particularly useful in handling disposal area problems include: Evaluation of Dredged Material Pollution Potential (Brannon, 1978), Confined Disposal Area Effluent and Leachate Control (Chen et al., 1978), Disposal Alternatives for Contaminated Dredged Material (Gambrell et al., 1978), and Upland and Wetland Habitat Development with Dredged Material: Ecological Considerations (Lunz et al., 1978).

7. Lake Conditions Most Suitable for Sediment Removal

Obviously, a lake with a filled basin might become a prime candidate for sediment removal. How much sediment is removed will depend not only on the amount of material obstructing use of the lake, but also upon the availability of suitable sites for disposal. The latter need not necessarily be a deterrent to sediment removal. There are various productive use options available for dredged sediment (Lunz et al., 1978; Walsh and Malkasian, 1978; Spaine et al., 1978). In the case of Nutting Lake, Massachusetts, where approximately 275,000 m^3 of sediment were removed, the material is being sold for \$1.00 m^{-3} effectively reducing dredged material removal cost to \$0.45 m^{-3} (Manfredonia, 1980). This not only solves the disposal problem, but reduces significantly the overall cost of the project

Due to the high cost of removing sediment from lakes, \$1.00 to \$15.70 m^{-3} as reported by Peterson (1979), it would seem that lake size might play an important role in determining lake types suitable for dredging. That does not appear to be the case. Surface area of lakes being dredged range from 2 ha to over 1,050 ha, and volumes being removed range from a few hundred m^3 to over 7 million m^3 (Peterson, 1979). While technically there is no reason a large lake cannot be dredged, there are some serious economic constraints. Most projects, however, have not been scrutinized closely from the economic standpoint.

Current information indicates that lakes with highly enriched surface sediment, relative to underlying sediment, would benefit from dredging (Andersson et al., 1975; Bengtsson et al., 1975). Lake Trummen, Sweden showed marked improvement when treated in this manner. Similar short term results were observed when 25-50 cm of organic sediment was removed from Steinmetz Lake, New York (Snow et al., 1980). This project differed significantly from that at Lake Trummen, however, since clean sand replaced almost all of the organic sediment removed (see case history #4 in this paper).

Sediment removal will not be cost effective where high sedimentation rates prevail. Sediment removal projects should have reasonable assurance of extended longevity. Generally, the smaller the ratio of watershed area to lake surface area, the greater the assurance of a lasting effect due to sediment removal. Watershed management practices should be encouraged with sediment removal projects where practical. However, massive watershed management programs to protect small lakes could be counter productive. Uttormark et al. (1974) have pointed out that where watershed area to lake surface area ratios greatly exceed 10 to 1, significant nutrient reduction from the watershed may be impractical.

Generally speaking, small lakes with organically rich sediment, low sedimentation rates and long hydraulic residence times would be ideal sediment removal candidates.

IV. Case Histories of Sediment Removal

Sixty-four pre-implementation, on-going, or completed sediment removal projects in the United States are summarized in the Appendix table. The table identifies sediment removal projects in 22 states. There are no doubt others which have not been included. Many lake sediment removal projects are undertaken by municipal districts, counties and even state organizations with little or no documentation of their effectiveness. The summary attests to this by the large number of data gaps. In most instances, where data do exist, they are not readily available. Published results are relatively scarce and documented long term (5 years or more) effects are simply nonexistent.

The five case histories presented were included for one or more of the following reasons: 1) they are examples of different sediment removal approaches, 2) they have different objectives, 3) they point out some strengths and weaknesses of the technique, 4) their results have been relatively well documented. The first, and perhaps best example of a well documented sediment removal project is located in Sweden.

1. Lake Trummen, Sweden

Lake Trummen, Sweden is one of the most thoroughly documented dredging projects in the world with both pre- and post-treatment information. This project represents an attempt to reduce internal nutrient cycling by skimming off a thin nutrient-rich sediment layer.

The lake has an area of 400 ha and, prior to restoration, a maximum depth of 2 m and a mean depth of 1.1 m. By the early 1960s, domestic waste discharge had deteriorated the lake to such a degree that the citizens of Växjo considered filling the basin (Bjork et al., 1971). Diversion of the major pollutant sources had little effect on the lake. Sediment removal implementation in 1970 was preceded by two years of study and project plan development. In 1970, one-half meter of finely divided organic sediment was dredged uniformly from the main lake basin (Bjork, 1974). In 1971, another half meter of sediment was removed bringing the total volume to about 400,000 m^3. This

deepened the lake by approximately 40 cm. Dredged material was disposed of partially in three macrophyte-infested, diked off bays. One large bay, representing approximatley one-third of the lake surface area and overgrown with macrophytes, was preserved in total as a wildlife habitat. Dredged material not placed in diked embayments was pumped to upland diked ponds where the return flows were treated with aluminum sulfate to reduce the phosphorus concentration in dredge slurry ($\sim$1 mg P liter^{-1}) to about 30 µg liter^{-1}. Dried dredge material was sold as top soil for about $2 (U.S.) m^{-3}. The total phosphorus concentration in the lake prior to sediment removal was 600 µg P liter^{-1}. Following sediment removal the concentrations occasionally have reached 70 to 110 µg P liter^{-1} (Bengtsson et al., 1975).

Bengtsson et al. (1975) indicated that phosphorus and nitrogen decreased drastically (Figure 5) and that the role of the sediment in recycling nutrients was minimized. The total and orthophosphate as well as the Kjeldahl-nitrogen levels have remained significantly lower than before or during the dredging program.

Biologically, the Shannon phytoplankton diversity index increased from 1.6 in 1968 to 3.0 in 1973 (Cronberg et al., 1975). Secchi disc transparency increased from 23 to 75 cm over the same period. Prerestoration mean annual phytoplankton productivity was 370 g C m^{-2} (1968-1969), declining to 225 g C m^{-2} following restoration (1972-1973). More than 60 percent of the annual phytoplankton production in 1972-1973 was attributed to algae less than 10 µm in size. Blue-green algal biomass was drastically reduced and some species disappeared, notably Oscillatoria agardhii.

Before dredging in Lake Trummen, the benthic fauna was dominated by oligochaetes and chironomids (Andersson et al., 1975). A year after dredging tubificid oligochaetes and chironomids became much more numerous. The total number of benthic organisms changed little. This was attributed to the mobility of chironomid larvae and the fact that the dominant species swarmed all summer, probably recolonizing newly dredged areas almost immediately.

The results at Lake Trummen are highly encouraging. Born (1979) indicated that today Lake Trummen is not only a usable recreational resource, but that the lake ecosystem has been restored. Figure 5 indicates that water quality has been remarkably good for four years following dredging compared to that prior to dredging. Jorgensen (1980), however, recently stated about Lake Trummen that, "a deterioration of the water quality has been recorded during the last 2-3 years (1976-1978), caused by the phosphorus input in the mixtures of rain and wastewater discharged to the lake during heavy rainfalls." No data were presented to determine what the nature of the deteriorated water quality might be.

2. Lake Herman, South Dakota

Lake Herman, South Dakota has a surface area of 546 ha, a maximum depth of 2.4 m and a mean depth of 1.7 m. The lake has a 145 km^2 agricultural watershed. Row crop and small grain farming practices beginning in the early 1900s have resulted in the deposition of approximately 2 m of silt over the entire lake basin. In 1970, 1971, and 1972, a 4.2 ha area of one bay in Lake

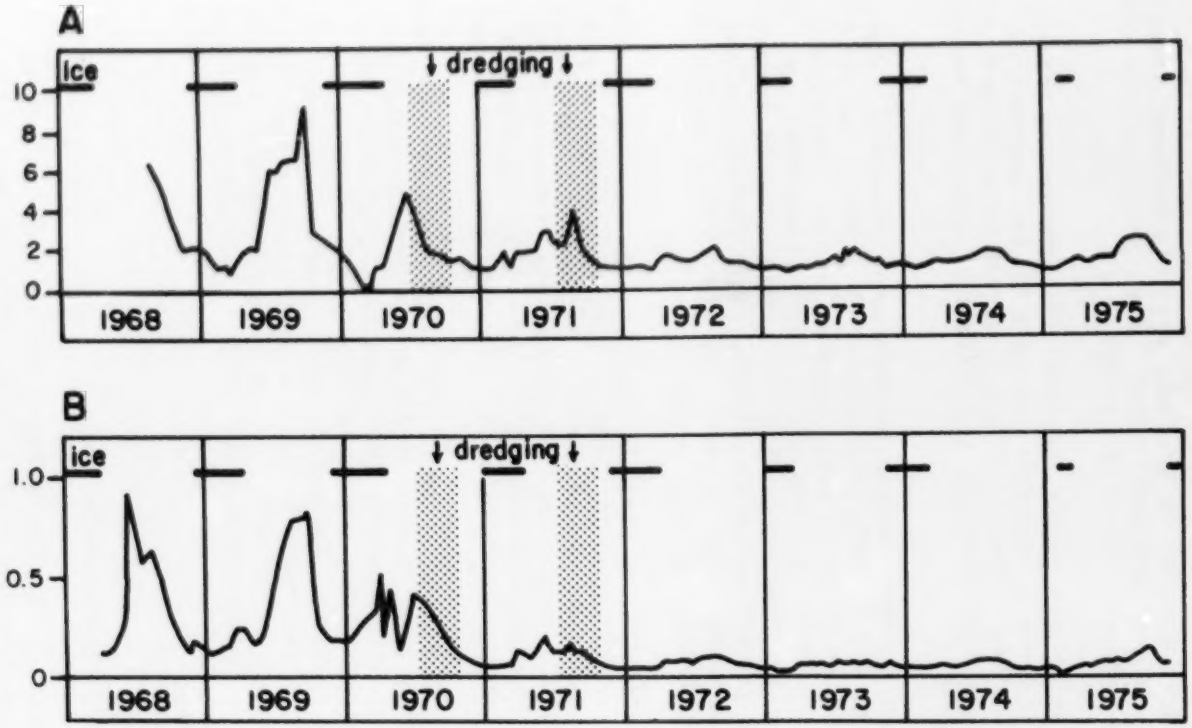
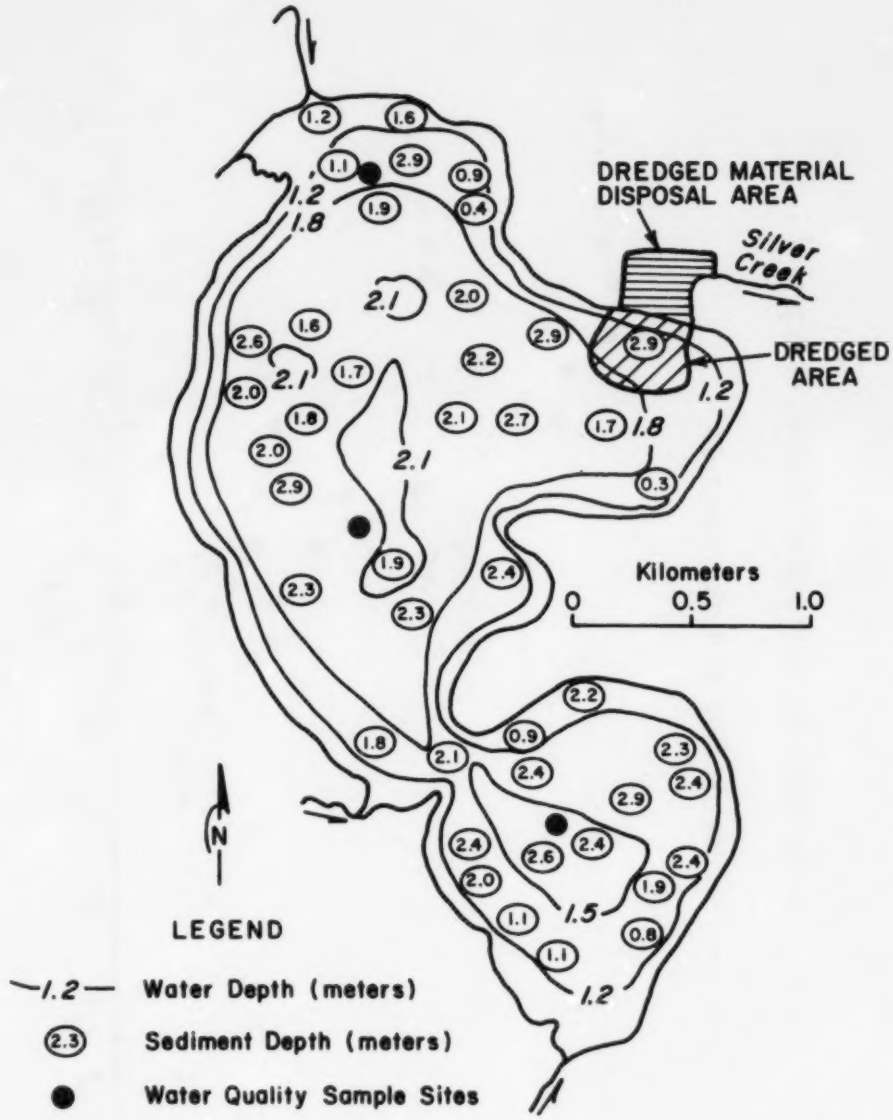


Figure 5. Lake Trummen. Organic nitrogen (A, mg N liter^{-1}) and total phosphorus (B, mg P liter^{-1}) in surface water 1968-1975. Data from W. Ripl. (modified from Bjork, 1978).



LAKE HERMAN, SOUTH DAKOTA

Figure 6. Lake Herman, South Dakota (redrawn from Churchill et al., 1975a).

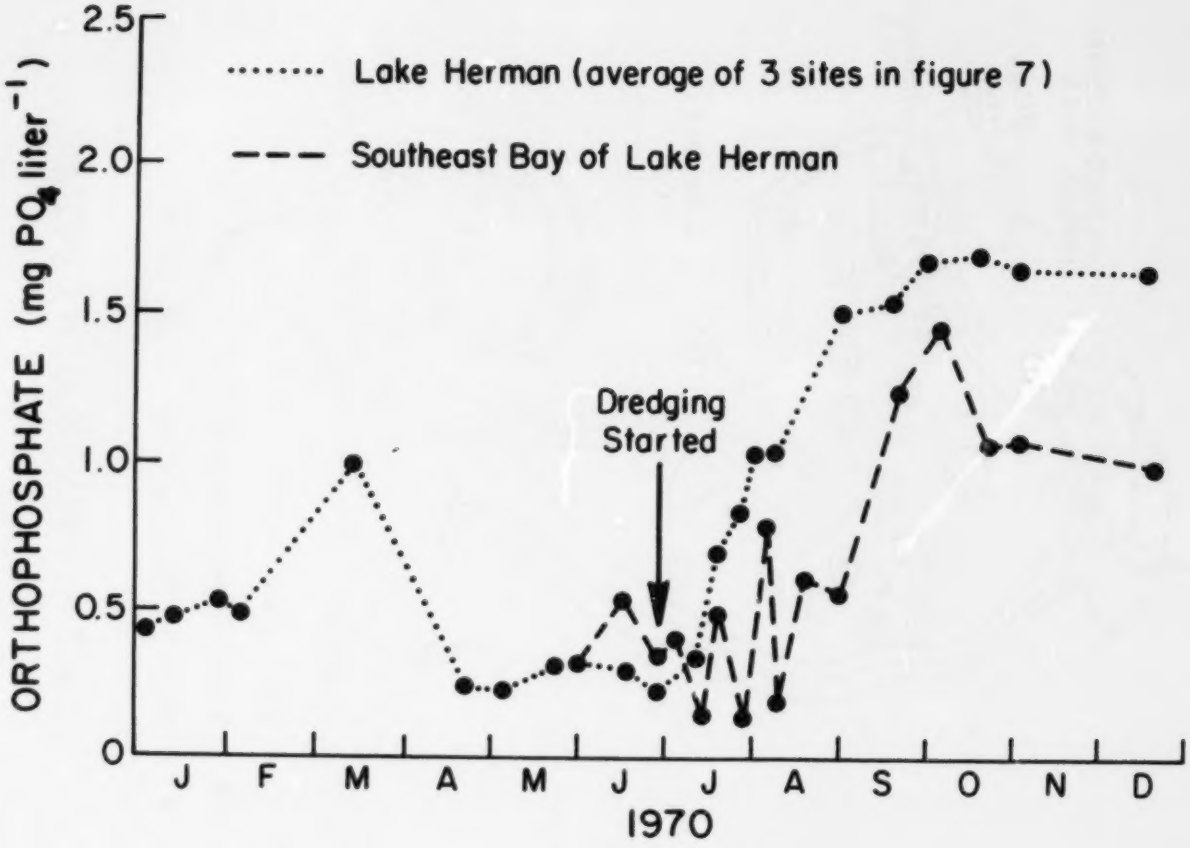


Figure 7. Orthophosphate (X 0.33 = P) in Lake Herman, South Dakota (modified from Brashier et al., 1973).

Herman was dredged. This work was undertaken to test the effect of hydraulically dredging nutrient-laden inorganic silt. This project serves as an example of a marginally successful, limited scale hydraulic dredging operation to reduce availability of nutrients in a shallow, intermittently destratified lake.

About 48,000 m^3 of silt were removed from the dredge site, deepening it from 1.7 m to approximately 3.4 m (Figure 6). Dredged material was deposited in a 3.4 ha area adjacent to the lake (Churchill et al., 1975a). At the end of the dredging project in 1972, drying had reduced the volume of the material about 50 percent. No coagulants were added to the dredge slurry.

Orthophosphorus concentrations in the lake increased dramatically from approximately 0.13 mg P $liter^{-1}$ to > 0.56 mg P $liter^{-1}$ shortly after dredging began (Brashier et al., 1973). Phosphorus concentrations in the southeast bay of the lake paralleled those of the main lake but appeared to lag behind by several days (Figure 7). Despite the rapid increase in phosphorus concentrations, they were not accompanied by major increases in phytoplankton productivity. This is not surprising, however, since Churchill et al. (1975b) indicated the lake was primarily nitrogen limited and that nitrogen frequently declined to zero during algal blooms. Orthophosphorus concentrations in Lake Herman prior to dredging were about 0.30 mg $liter^{-1}$, during dredging (summer 1970, 1971, and 1972) they were almost uniformly 0.17 to 0.50 mg $liter^{-1}$, while after dredging (summer 1973, 1974, and 1975) they ranged form 0.10 to 0.50 mg $liter^{-1}$ with a tendency toward the lower end of the range (Churchill et al., 1975b). Churchill and associates were reluctant to conclude that dredging was responsible for the increase in phosphorus concentration in Lake Herman, however, they indicated that "there were no other noticeable environmental changes that could readily account for this dramatic increase in phosphates". While no apparent increases in phytoplankton production resulted from the increased phosphorus concentrations, the same may not have been true if the lake was not normally nitrogen limited. If the large increases in phosphorus were actually attributable to dredging one might conclude that this particular dredging operation was inefficient in terms of limiting resuspension of sedimented material. While the project demonstrated that nutrient-laden silt could be removed by hydraulic dredging, thereby reducing the amount available for internal cycling in the area dredged, it left a question about the efficiency of such operations.

3. Wisconsin Spring Ponds

The Wisconsin Spring Ponds (Sunshine and Krause Spring Ponds) Project represents another of the more thoroughly documented records concerning the ecological effects of dredging small lakes. The purpose of the dredging was to deepen the ponds and increase their area to improve fish production. Incidental to the deepening was the control of aquatic macrophytes. This was an important plus for the project since Carline and Brynildson (1977) stated the belief that aquatic macrophytes played an important, if not the most important role, in determining the rate of filling in spring ponds. Presumably this conclusion might have implications for other types of small closed basin lakes such as Lilly Lake, Wisconsin (Wis. Dept. Nat. Resour., 1969).

The physical characteristics of the two spring ponds before and after dredging are shown in Table 3. Sediment removal from the entire basin represents the most severe short run environmental impact possible. It completely disrupts the benthic invertebrate population and removes almost all aquatic macrophytes. In Sunshine Pond the predredging mean annual density of benthos was 5,500 organisms m^{-2}. Immediately after dredging the number was reduced to 84 m^{-2}. Five years after dredging the density of benthic invertebrates had increased to 16,000 m^{-2} in Sunshine Pond. Results for Krause Pond were similar. However, while most of the invertebrates prior to dredging were chironomid larvae, those five years later were primarily oligocheates which are not an important trout food.

Carline and Brynildson (1977) indicated that after four years the important fish food organisms, such as amphipods and chironomid larvae were still in the process of reestablishing. This had a temporary, but statistically insignificant, effect on fish production. Fish growth rates declined sharply in 1971 during the dredging operation but by 1975 had returned to predredging values.

Chara (a non-rooted, non-vascular aquatic macrophyte) covered 60 percent of the Sunshine Pond basin prior to dredging. Dredging removed essentially all of the plant material. Five years after dredging Chara was reestablished over 28 percent of the pond. However, its biomass was only about 10 percent of the predredging level.

Lake dredging frequently is of concern to fisheries managers because of the suspected adverse impacts on fish populations. Destruction of fish food organisms is often the focal point of this concern. The above indicates that benthic invertebrate reductions had but a limited impact on the fishery in the short run. Carline and Brynildson (1977) indicated that four to five years after restoration by dredging the average density and biomass of fishable-size fish were substantially greater than during the pre-dredging period.

4. Steinmetz Lake, New York

This case history is presented because of its relatively unique whole-lake treatment technique, which consisted of complete drawdown, bulldozing out of approximately 25 to 50 cm of organic sediment and replacement of the sediment with clean quarry sand. Macrophyte control and beach improvement were the objectives.

Records indicate Steinmetz Lake, in Schenectady, New York was once a quarry (Snow et al., 1980). The city acquired the lake and surrounding property for a park in 1935. In 1936, the lake was drained and some deep holes filled to improve swimming. It has been maintained by the city since that time for recreational use.

The lake is 1.2 ha in area, with a mean depth of about 1.5 m and a maximum depth of 2.1 m. It has a 28-40 ha naturally vegetated watershed which serves as a public park (Snow et al., 1980). Day use of the swimming and park facilities declined rapidly since 1971 (over 17,000 users) to a low of approximately 6,000 in 1975. The decline was attributed to excessive weed growth and

turbidity due to algal production, caused by nutrient loading from stormwater drains and surface runoff, lawn fertilizing in the watershed, and the in-lake macrophyte growth/death/decay cycle which was rapidly building up the organic sediment (Snow et al., 1980).

The restoration project was implemented in 1978. It included sediment removal, diversion of stormwater drains, and the construction of a surface runoff interceptor drains. Bulldozing 2026 m^3 of sediment required much more time than anticipated (six weeks) due to sinking of equipment in the soft sediment (Snow et al., 1980). Backfilling with clean sand was much less of a problem. About 1843 m^3 of sand was placed in eight days. This technique results in no net changes of lake bathymetry, but produces a completely new, clean substrate.

The short term results of the Steinmetz Project have been dramatic. Snow et al. (1980) and Bloomfield (1980) indicate the average summer Secchi disc readings have improved from the 1976 and 1977 readings of 1.25 m and 1.33 m, respectively to post-treatment readings in 1978 and 1979, of 1.47 m and 2.20 m, (maximum depth of the lake). Average summer chlorophyll a levels in 1976 and 1977 were 10.4 µg liter^{-1} and 5.8 µg liter^{-1}, respectively. Post-treatment levels have been reduced to 0.1 µg liter^{-1} in 1978 and 1.8 µg liter^{-1} in 1979. Aquatic macrophyte biomass, primarily Potamaogeton crispus, ranged from 30-50 g wet weight m^{-2} before sediment removal and was reduced to virtually zero after treatment. Snow et al. (1980) reported that after sediment removal P. crispus was observed growing at irregular intervals in straight lines at a density of 2 to 20 plants m^{-2}. Closer observation revealed that the plants were growing where tracked vehicles had forced incompletely removed organic sediment up through the sand cover during the sand spreading operation. Snow et al. (1980) indicated that the extruded organic sediment was acting as a rooting media for P. crispus.

Snow and associates (1980) rated the project as successful. The public is once more enjoying the recreational use of the lake. In 1977, use was virtually zero. In 1978, 2,050 people used the beach area. In 1979 the number increased to 3,285.

5. Lilly Lake, Wisconsin

Lilly Lake, located in southeastern Wisconsin (Kenosha County), has an area of 37 ha and a watershed of 155 ha. The lake suffered from a severe shoaling problem, a mean depth of 1.4 m and a maximum depth of 1.8 m. Dunst (1980b) indicated the lake basin was filled with 10.7 m of sediment containing a high proportion of non-decomposed plant material.

A report by the Wisconsin Department of Natural Resources (1969) indicated the main problem in Lilly Lake was its shallow depth and abundant growth of aquatic plants over the entire lake basin. No indication of plant density was mentioned. However, it was noted that mechanical removal of aquatic vegetation had taken place for several years in an attempt to maintain an area suitable for boating and swimming. The DNR report indicated P. amplifolius was the dominant plant in the lake and that spring total phosphate concentration at a depth of 1 m in 1966 was 0.26 mg liter^{-1} (0.09 µg P liter^{-1}). The dissolved phosphate concentration was 0.04 µg liter^{-1} (0.01 µg P liter^{-1}).

The fish population was dominated by slow growing yellow perch. Chemical eradication and restocking with largemouth bass, sunfish and northern pike was generally unsuccessful (Wisconsin DNR, 1969). Winter fish kills were common. It was concluded that deeper water was required to manage for fish production. The DNR recommended that at least 10 percent of the lake be dredged to a depth of approximately 6.0 m. There is no indication of how that particular depth or area was determined.

No action was taken on the DNR recommendation until 1976 when Wisconsin received a grant from the U.S. Environmental Protection Agency for the purpose of restoring Lilly Lake. The restoration plan submitted by the State called for dredging about 665 x 10^3 m^3 of sediment from the lake, deepening it to a maximum depth of about 6 m. The objective of the project was to deepen the lake sufficiently so that plant growth and winter fish kills would no longer be serious problems (Wisconsin DNR, 1975). The project proposal called for disposal of most of the dredge material in a nearby abandoned gravel pit.

Pre-implementation indications were that the sediment would flow to the intake of a suction dredge and that cutterhead swing could be eliminated. However, when dredging began in July 1978 it was quickly discovered that the non-decomposed plant material in the sediment formed a tight matrix which resisted flowing. From early July to October 26, 1978, approximately 360 x 10^3 m^3 of sediment were pumped from the lake (Dunst and Beauheim, 1979). During the 1978 dredging period in-lake 5-day BOD ran about 1-2 mg O_2 $liter^{-1}$ greater than during the previous year. Turbidity increased in the same manner from 1 to 3 formazin units. Ammonia concentration increased dramatically from approximately 0.01 mg $liter^{-1}$ during most of 1977 to a high of nearly 5.5 mg $liter^{-1}$ when dredging ceased in October 1978. Although predredging records of particulate phosphorus concentrations were incomplete it appeared that there was a steady increase during dredging. Chlorophyll *a* levels in the lake during 1977 were steady at about 2.5 to 3.0 µg $liter^{-1}$. In 1978 chlorophyll *a* reached a high of approximately 27 µg $liter^{-1}$ immediately after dredging began and then decreased to a level ranging from 12 to 18 µg $liter^{-1}$ for the remainder of the dredging season. Dunst and Beauheim (1979) reported that while productivity had averaged about 200 mg C m^{-3} day^{-1} during the summer of 1976 and 1977, it increased to an average of about 750 mg C m^{-3} day^{-1} in 1978.

Dredging commenced again in May 1979 and was completed by September (Dunst, 1980b). Sediment removal actually exceeded 680 x 10^3 m^3 and the lake now has a maximum depth of 6.5 m. Additional Lilly Lake dredging data have been reported by Dunst (1980c). The summer of 1980 was the first full growing season since dredging. Dunst (1980a) has indicated that in-lake macrophyte biomass of 200 to 300 g dry wt m^{-2} in 1977 was reduced to virtually zero when dredging ceased in September 1979. Water quality since that time has remained good and local sponsors are generally pleased with the outcome (Dunst and Beauheim, 1980). It will be interesting to observe over the next few years if macrophytes and internal nutrient cycling remain diminished as a result of the lake deepening at Lilly Lake.

V. Costs of Sediment Removal

Sediment removal costs vary widely depending on type and quantity of sediment being removed; type of removal method, i.e., bulldozer, mechanical hydraulic or special purpose dredge, sediment transport distance and method of transporting; availability of disposal area; and ultimate use of dredged material. Even within the same geographic region using the same dredging equipment, costs may vary widely. Carline and Brynildson (1977) indicated that cost per cubic meter of sediment removal to increase the volume of spring ponds in Wisconsin ranged from $0.52 to $2.67. On a national basis the cost differentials are much greater (Table 3). The Army Corps of Engineers has shown a similar broad range in sediment removal costs on a national basis (Saucier, 1976).

Generally speaking, per unit volume costs for sediment removal will be inversely related to the total volume of material removed when disposal is not a problem. Contaminated dredged material, requiring treatment and perhaps upland containment, may increase cost by 3 to 5 times (Peterson, 1979). On the other hand, sale of "desirable" dredged material may help offset the removal cost. Nutting Lake, Massachusetts sediment is being sold for $1.40 m^{-3} (Manfredonia, 1980). A portion of the sediment from Lake Trummen, Sweden was sold for nearly $2.00 m^{-3}. Innovative disposal practices may significantly reduce the cost of sediment removal. Closer examination of cost figures is required before reliable generalizations can be stated.

VI. Summary

1. The purposes for removing sediment from a lake are 1) to deepen it for improved boating, fishing, swimming and other recreational activities, 2) to prevent or reduce the internal sediment/water recycling of nutrients, 3) to reduce the effects of toxicants, and 4) to remove and reduce the growth of nuisance aquatic plants. The expected results of these treatments are improved dissolved oxygen concentrations; less sediment water nutrient exchange, accompanied by reduced algal biomasses, increased clarity, and reduced macrophyte nuisance problems. All of these goals may not be attained in all cases.

2. The technique is effective for deepening, and when a lake has shoaled extensively there may be no other practical method for restoration. Sediment removal for deepening is accompanied by nutrient removal which no doubt contributes to the overall improved lake quality in many sediment removal projects.

The duration of deepening effectiveness will depend largely on lake sedimentation rates. The one case history of skimming nutrient-rich surface sediments for controlling internal nutrient cycling has been dramatically successful for approximately six years. Signs of reduced water quality after that time were associated with external nutrient loading, however, water quality has remained better than it was prior to dredging.

Macrophyte control by sediment removal is usually attempted through deepening. The effectiveness of this approach is not clear at this point. Sediment texture, nutrient content, and water temperature may be as important as light level in determining if plants reinvade a dredged area.

3. The reduction of toxic problems in lakes by sediment removal is technically feasible. This technique is being practiced at a high level of sophistication in Japan. It requires special dredging equipment, confined disposal areas and treatment of dredged material return flow waters. Costs are usually double to quadruple those for removing uncontaminated sediment.

4. Relatively unsuccessful sediment removal projects can usually be associated with extremely limited sediment removal (simply too small a scale to be effective), an inefficient dredging operation (poor production of solids in the dredge slurry, excessive downtime, extreme turbidity problems or inadequate disposal areas), or no attempt to control external loading of pollutants to the lake.

5. Any project considered for sediment removal should be preceded by an investigation of nutrient and sediment mass balances. Mass balance calculation will provide information on the relative proportion of pollutants derived from the various sources, thus providing a mechanism to focus on the critical compartments. Accurate portrayal of sedimentation rates permit the extrapolation of sediment removal longevity, thus the comparative cost effectiveness of sediment removal.

6. There are a number of environmental concerns associated with sediment removal activities. The majority of those associated with the sediment removal phase (in-lake) appear to be of short duration and thus limited significance when compared to the longer range effectiveness of the treatment. This may not be the case when toxic substances are involved if sediment removal is poorly managed. The effects of dredged material disposal may be longer lasting. These effects will be tied directly to the method of disposal and the quality of the dredged material. Filling of wetlands (no longer commonly accepted) results in permanent destruction of that habitat. Enrichment of agricultural land with non-toxic dredge material may have a long range positive effect. Imaginative disposal plans can reduce the impact of disposal and the overall cost of a dredging project.

7. The hydraulic dredge is the preferred piece of equipment for most sediment removal operations. It is comparatively efficient in terms of production and relatively clean (resuspension of sediments) in its operating cycle. Special purpose dredges reduce resuspension of sediment, but are relatively inefficient in terms of energy expended to remove a given volume of sediment. Therefore, operational costs are usually higher than for conventional hydraulic dredges. Their use may be warranted only when toxic substances are being removed.

8. An average unit cost for sediment removal cannot be stated. The variables associated with dredging are so numerous that an average unit cost figure becomes meaningless. About the only generalized statement that can be made about unit cost is that it is likely to be inversely proportional to the amount of sediment removed.

9. There are still a number of uncertainties associated with dredging as a lake restoration technique. Among these are the longevity of effectiveness, the relative effectiveness of partial compared to whole-lake dredging, how to dredge in order to minimize ecosystem shock and encourage rapid repopulation of desirable organisms. In cases where nutrient control is of major interest, more experience is needed with surface sediment skimming compared to deepening as a restoration technique. A clearer picture of the comparative effectiveness of these two techniques could have a major influence on future projects. Costs could be reduced significantly if it was demonstrated that surface skimming of sediments might accomplish the same objectives as deepening. More sustained research and monitoring of the effectiveness of diverse sediment removal methods is required.

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APPENDIX TABLE
Status of various lake sediment removal projects in the United States.†

Ref[1]	Location and Lake	Objective[2]		Pretreatment Chem, Phys, Biol Data[4]	Current Project Status[3]								Other Concurrent Treatment
		Nutr. Cntrl.	Deepen		Sediment Removal Method[5]	Volume Sediment Removed ($m^3 \times 10^3$)	% Lake Basin Dredged	Post-treatment Chem, Phys, Biol Data[4]	Cost[6]	Sediment Removal Date(s)	Duration of Effectiveness[7]	Side Effects Demonstrated	
	CALIFORNIA												
LR, GA	Ellis	X		A_o=12.5 $\bar{z}$=1.5 J=0.003	?	.002	100	?	?	79	S	Recreational Benefits	Partial Storm Sewer Diversion, Herbicides for Hydrilla
LR, GA	Gibraltar	X	X	A_o=110.8 $\bar{z}$=13.4 wm monomictic	4	306	Small	0	tot=*2 mill. Pilot Project	80	0	N	Negotiating Land Use Practices w/U.S. Forest Service, Sediment Hg Removal
LR, GA	Stafford	X		A_o=80.9 z_m=10.7 (TP≅150) (TN≅2050) A_d≅2150	4	*76	25 Shallow Areas Only	0	tot= *220K	80	0	Park Enlarged with Dredged Material	Buffer Zone, Erosion Control
LR, GA	Temescal	X		A_o=5.78 $\bar{z}$=3.2 z_m=5.8 V_d=1.9E5 SD≅2.3 TP≅250 TN≅1135	3	36	~50	z_m=26.1	tot= 609K	79	S	Improved Fishery	Source Control Program
	COLORADO												
D, GA	Ladora	X		A_o=24	?	Upper 15 cm	?	?	tot= 18K First Phase	60-66	Still Some Dieldrin Present in Fish	N	Sedimented Dieldrin Removal

†Footnotes are listed at end of table on page 55.

(continued)

Ref[1]	Location and Lake	Objective[2]		Pretreatment Chem, Phys, Biol Data[4]	Current Project Status[3]								Other Concurrent Treatment
		Nutr. Control	Deepen		Sediment Removal Method[5]	Volume Sediment Removed ($m^3 \times 10^3$)	% Lake Basin Dredged	Post-treatment Chem, Phys, Biol Data[4]	Cost[6]	Sediment Removal Date(s)	Duration of Effectiveness[7]	Side Effects Demonstrated	
	FLORIDA												
GA	Eola	X		A_o=11 Ad=71.3 z_m=7.6 z=35 V=4.1E5 J=0.64 SD~1.1 TP~120 TNO_2~100	2 Consolidation, Sand Cover	38	~50	?	~1.00	73	S 4-5	N	Storm Sewer Trash Traps
	ILLINOIS												
D	Carlinville	X		A_o=68	1a	83	?	?	1.07	68-71	?	Spoils Site Produced Bumper Crop of Wheat	?
LR, GA	Frank Holten[c] (3 lakes)		X	A_o max = 1.6, 16.5, 21.4 z_m=5.18, 10.7, 2.74	1	$411*	Lake % 3 100 2 30	0	?	79 Pilot	0	0	Stream Diversion
D	Oakland		X	A_o=1050 z_m=4.5	?	23	?	?	~0.65	72	?	?	?
	IOWA												
LR	Blue	X		A_o=371.5 z_m=4.9	1a	$306*	36	*	?	*81	0	*	Water Treatment

(continued)

Ref[1]	Location and Lake	Objective[2]		Pretreatment Chem, Phys, Biol Data[4]	Current Project Status[3]								Other Concurrent Treatment
		Nutr. Cntrl.	Deepen		Sediment Removal Method[5]	Volume Sediment Removed ($m^3 \times 10^3$)	% Lake Basin Dredged	Post-treatment Chem, Phys, Biol Data[4]	Cost[6]	Sediment Removal Date(s)	Duration of Effectiveness[7]	Side Effects Demonstrated	
	IOWA (Continued)												
GA	Lenox	X		A_o=13.4 z_m=3.4 z=.9	1	76	100	0	2.35	?	0	0	Diking
GA	Oelwein	X		A_o=16.18 z=1.82 SD~0.9	1	266	?	?	tot=30K	?	?	?	Reconstruct Siltation Basin
	LOUISIANA												
LR	City Lakes (5)	X		A_o=2.5-83.4 z=0.5-1.5 J≅.15-.4 z_m=0.7-1.6 A_d=1.6-299	*1	*	?	?	tot=*3 mill.	*	*	*	Sump, Land Use Practices
	MAINE												
D, GA	Sebasticook	X		A_o=1740 z_m=17.7 SD<2 TP=15-140 V=11.0E6 J=0.7	*3	*	<1	?	tot=58K	81	*	*	Wastewater Treatment, Rebuild dam, Drawdown
	MASSACHUSETTS												
LR, GA	Cochituate	X		A_o=248.5 z_m=13.1 SD≅0.8	2 Consolidation Only	* 26,761 From Settling Ponds	*	*	* Project Not Finalized	*	*	*	Dredging of Settling Ponds, Rough Fish Harvest, Natural Sand Filter Beds

(continued)

Ref[1]	Location and Lake	Objective[2]		Pretreatment Chem, Phys, Biol Data[4]	Current Project Status[3]								Other Concurrent Treatment
		Nutr. Cntrl.	Deepen		Sediment Removal Method[5]	Volume Sediment Removed ($m^3 \times 10^3$)	% Lake Basin Dredged	Post-treatment Chem, Phys, Biol Data[4]	Cost[6]	Sediment Removal Date(s)	Duration of Effectiveness[7]	Side Effects Demonstrated	
	MASSACHUSETTS (Continued)												
LR	Ellis Brett	X Weed Control	X	A_o=2.02 z_m=3	?	?	?	?	tot= *213K	?	0	Recreational Benefits	Streetsweeping, Filters, Catch Basins
LR, GA	Morses Pond	X		A_o=49.4 $\bar{z}$=3	1	199	22.1	?	1.56	77	?		Harvest, Replace Deciduous Trees, Chemical Removal of Iron and Colloids, Wetlands Buffer Zone
LR, GA	Nutting	X		A_o=31.6 SD~1.07 z=1.28 TN~200-2280 z_m=2.1 J=.74 TP=20-190	1	275	56.3	?	1.22	78-present	0	Sale of dredged material for $1.40/$m^3$	Post-Dredge Flocculation, Stormwater Diversion, Buffer Zones, Sediment Entrapments
D, GA	Pomps		X	A_o=8 z_m=3	X 2 Some Replacement w/Sand	*2	~10 Bathing Area Only	?	tot= 6-10K	73	S 5-6	Improved Recreation	Water Level Stabilization by Control Structures
	MICHIGAN												
D, GA	Long		X	A_o=59 z_m=2.1	X 1a	~810		A_o=63 $\bar{z}$=2 z_m=4	0.22	61-65	S; No Winter Kills	Increased Property Values; Fish Condition Improved	None Mentioned

(continued)

Ref[1]	Location and Lake	Objectives[2]		Pretreatment Chem, Phys, Biol Data[4]	Current Project Status[3]							Other Concurrent Treatment
		Nutr. Cntrl.	Deepen		Sediment Removal Method[5]	Volume Sediment Removed ($m^3 \times 10^3$)	% Lake Basin Dredged	Post-treatment Chem, Phys, Biol Data[4]	Cost[6]	Sediment Removal Date(s)	Duration of Effectiveness[7]	Side Effects Demonstrated	
	MICHIGAN (Continued)												
LR, GA	Lansing ($NaAsO_3$ Problem)	X Weed Control	X	A_o=182 $\bar{z}$=1.5 z_m=9 J=1.6 TP~50 TN~170 SO~1.8	1	1,230	64	$\bar{z}$=*3	1.00	79-present	0	Improved Recreation, Potential Adverse Impact on Pike Fishing	Erosion Control of Spoils; Beach Nourishment Creation of Pike Spawning Area
	MINNESOTA												
LR, GA	Albert Lea	X		A_o=1012 $\bar{z}$=1.2 z_m=4.1 J≅0.2 SD≅0.18 TP≅300 TN≅300	2	*84	0	*3	?	*	*	*	Education, Land Use Practices, 3° Sewage Treatment
D	Crystal	X		A_o=160 z_m=4.5	?	?	?	A_o=160 z_m=4.5	?	70	?	Economic Value of Dredged Material, Sediment Input Negated Dredging	?
LR, GA	Long	X		A_o=74.5 $\bar{z}$=3.7 z_m=8	1	217	?	0	2.96	?	0	0	Erosion Control, Wetland "Filter", Creek Diversions

(continued)

Ref[1]	Location and Lake	Objective[2]		Pretreatment Chem, Phys, Biol Data[4]	Current Project Status[3]							Side Effects Demonstrated	Other Concurrent Treatment
		Nutr. Cntrl.	Deepen		Sediment Removal Method[5]	Volume Sediment Removed ($m^3 \times 10^3$)	% Lake Basin Dredged	Post-treatment Chem, Phys, Biol Data[4]	Cost[6]	Sediment Removal Date(s)	Duration of Effectiveness[7]		
	MINNESOTA (Continued)												
LR	Penn	X		A_o=14.6 z_m=1.1 TP≅275	?	?	?	?	?	?	?	?	Aeration, Sedimentation Basins
	MISSOURI												
D	Big	X		A_o=250 z_m=1.2	1a	420	17.5	z_m=2.4	0.06/m^3/yr (disposal	66	?	?	Levees to Prevent
of spoils-					Floodwater				no cost)				Intrusion
LR	Creve coeur	X		A_o=148 $\bar{z}$=0.5	1a	?	82	z_m=3	*0.80/m	*73-77	?	Increased Recreation Benefits	
D	Lion's	X		A_o=2.4	2	61	"most"	z_m=5	0.16/m^3	69	?	Good Fishing, Enhanced Recreational Value, No Weed Problem	Upstream Modifications to Reduce Siltation
LR	Vandalia	X		A=15.4 $\bar{z}$=3.9	?	*137	?	?	?	*	?	?	Sediment Catchment Basins

(continued)

Ref[1]	Location and Lake	Objective[2]		Pretreatment Chem, Phys, Biol Data[4]	Current Project Status[3]								
		Nutr. Cntrl.	Deepen		Sediment Removal Method[5]	Volume Sediment Removed ($m^3 \times 10^3$)	% Lake Basin Dredged	Post-treatment Chem, Phys, Biol Data[4]	Cost[6]	Sediment Removal Date(s)	Duration of Effectiveness[7]	Side Effects Demonstrated	Other Concurrent Treatment
	MONTANA												
GA	Sacajawea		X	$\bar{z}$=.5	2	?	100	?	tot= *464K	80	*	N	Flushing w/ River Water
	NEW JERSEY												
D	Brainard	X	X	A_o=5 z_m-1.8 1	?	?	?	z_m=2.7	1.30	73	?	N	
D, GA	Carnegie		X	A_o=110 z_m=3	1a 3 (Select Areas)	765	>75	Increased $\bar{z}$	1.30	72-73	S to Present	Creation of Marsh for Wildlife with dredged materials	
D	Plainsboro		X	A_o=14 z_m=1	3 (drag)	?	?	?	tot= $145,166	72	1 yr	Creation of Parks w/Dredged Material	
	NEW YORK												
LR, GA	Buckingham	X		A_o=1.82 z_m=1.65	2	?	?	$\bar{z}$=1.0	tot= 16.5K	?	Not Highly Effective Still has Storm Sewer Inflow	?	?
LR, GA	Collins Park	X	X	A_o=24.3 z_m=9.75 $\underline{V}$=6.5E5 $\bar{z}$=2.7 J=0.25	1	52	15	A_o=23.84 z_m=9.14 $\bar{z}$=2.9 V=6.8 E5	2.75	78-79	S "limited" to 15% Area Dredged Where *P. Crispus* was Reduced by 95%.	Dredging Limited by Discovery of Paleo-Indian Artifacts. A 0.6 m Lake Drawdown.	Planting Aquatics to act as Nut. Trap Watershed Cleanup

(continued)

Ref[1]	Location and Lake	Objective[2]		Pretreatment Chem, Phys, Biol Data[4]	Current Project Status[3]								Other Concurrent Treatment
		Nutr. Cntrl.	Deepen		Sediment Removal Method[5]	Volume Sediment Removed ($m^3 \times 10^3$)	% Lake Basin Dredged	Post-treatment Chem, Phys, Biol Data[4]	Cost[6]	Sediment Removal Date(s)	Duration of Effectiveness[7]	Side Effects Demonstrated	
	NEW YORK (Continued)												
LR	Delaware Park	Decontamination		A_o=11.9 $\bar{z}$=1.1 z_m=3.2 TP≅45 TN≅200 SD≅0.5	*	*	*	*	tot= *1 mill *	*82	*	*	Diverson of Polluted Stream
LR, GA	Hampton Manor	X		A_o=3.2 $\bar{z}$=2.1	2 Changed to 1; "mud cat"	23	?	SD≅1.5	tot=100K	77	"to present'	Fish in Better Condition	Aeration, Clean-flo, Storm Sewer Repair
LR, GA	Hyde Park	X Decontamination		A_o=12.95 $\bar{z}$=0.6	2	*	*	*	* Out to Bid	*	*	*	Siltation Ponds, Oil Boom, Erosion Control
LR, GA	Tivoli	X		?	2	30	100	0	0	?	0	0	Covered Bottom w/ Quarry Sand
LR	Steinmetz	X	X	P crispus 40 g/m² A_o=1.6 $\bar{z}$=2.4 z_m=6.1 V=8.1E5 SD~1.25 CA≅5-10µg/l	2	.003, Replaced w/Quarry Sand	75	P. Crispus Absent CA≅<0.2µg/l SD~Bottom $\bar{z}$=2.4 z_m=6.1	Sediment Removal= 4.34 Sand Replacements= 13.34	78	Generally Effective	Suitable for Bathing	Seal Nearshore Sediments w/Sand

(continued)

Ref[1]	Location and Lake	Objective[2]		Pretreatment Chem, Phys, Biol Data[4]	Current Project Status[3]								Other Concurrent Treatment
		Nutr. Cntrl.	Deepen		Sediment Removal Method[5]	Volume Sediment Removed ($m^3 \times 10^3$)	% Lake Basin Dredged	Post-treatment Chem, Phys, Biol Data[4]	Cost[6]	Sediment Removal Date(s)	Duration of Effectiveness[7]	Side Effects Demonstrated	
	NEW YORK (Continued)												
LR, GA	Washington Park	X		A_o=2.3 z_m=3.5 V=3.7E4	2	?	?	$\bar{z}$=2.0 V=4.5E4 Increased SD	tot=45K	?	Cleaner w/ High Nutrient Conc.	No More Weed Problems	
LR	59th St. Pond (Central Park)	X Decontamination	X	A_o=1.80 $\bar{z}$=0.15	2	13	100	?	13.93	78	S	*	Clean Storm Drainage Pipes Shoreline Repair
	NORTH CAROLINA												
LR	Mystic	X		A_o=4.0 z_m=4.9 $\bar{z}$=1.4 J=0.08	1,2	?	100	?	tot=55K	77-80-Sed. Ponds Only	S	?	Shoreline Repair, Sediment Control Dams
	OKLAHOMA												
GA	Sunset Lake			A_o=42	X 2	*382	*	*	?	80	*	*	Best Land Use Practices
	OREGON												
GA	Commonwealth			A_o=2.6 $\bar{z}$=0.9	X 1,2, 3(drag)	19	100	z_m=1.8	tot= 153.4K Dredge= 15K	78	S, to Present	Increased Property Values	Shoreline Stabilization "Periodic" Chemical Treatment for Water Milfoil; Landscaping at Park

(continued)

Ref[1]	Location and Lake	Objective[2]		Pretreatment Chem, Phys, Biol Data[4]	Current Project Status[3]								
		Nutr. Cntrl.	Deepen		Sediment Removal Method[5]	Volume Sediment Removed ($m^3 \times 10^3$)	% Lake Basin Dredged	Post-treatment Chem, Phys, Biol Data[4]	Cost[6]	Sediment Removal Date(s)	Duration of Effectiveness[7]	Side Effects Demonstrated	Other Concurrent Treatment
	SOUTH DAKOTA												
GA	Covell	X		A_o=4.15 $\bar{z}$=2.0 TP≅500 TN~2000	1	?	100	z_m=*4.6	tot= *50K	80	?	Creation of Park	Siltation Basin, Some Storm Sewer Diversion
D, GA	Herman	X		A_o=5.30 z_m=3	1a	54	1	*	*0.25	*	*	Increased Fertility at Disposal Site; Inc. TP Hardness, Silica when Dredging Began; Turbid	*
	TEXAS												
GA	McQueeney	X		A_o=166 z_m-12 J≅0.006	*1	80	2.5	?	tot= 240K	79 to Present	?	Unresolved Septic Tank Problems	Watershed Management
D	White Rock	X	X	A_o=280 z_m=7.3	*1	*	*33	*	*	*	*	Property Owners in Spoils Area Creating Problems	?
	VIRGINIA												
D	Barcroft	X		A_o=55 Sediment Control	?	.004/yr	?	?	tot= 300K	60-70	?	?	Land Use, Improved Drainage System

(continued)

Ref[1]	Location and Lake	Objective[2]		Pretreatment Chem, Phys, Biol Data[4]	Current Project Status[3]								Other Concurrent Treatment
		Nutr. Cntrl.	Deepen		Sediment Removal Method[5]	Volume Sediment Removed ($m^3 \times 10^3$)	% Lake Basin Dredged	Post-treatment Chem, Phys, Biol Data[4]	Cost[6]	Sediment Removal Date(s)	Duration of Effectiveness[7]	Side Effects Demonstrated	
	WASHINGTON												
D, GA	Capitol	X	X	A_o=120 z_m=6.1	1a	765	30	?	?	79	S	?	Sedimentation Basins; Park in Sports Area
LR, GA	Liberty	X Weed Control	X	A_o=316 $\bar{z}$=7 z_m=9.1 Ad=3444.6	1	*	40	*	tot= 1.2 mill.	80	*	8	Flood, Water Supply Control for Adjacent Wetland
LR, GA	Long	X		A_o=137 z_m=3.7 Polymictic	1	46	5	?	tot=489K	78	S	?	Drawdown, Chemical Treatment, Settling ponds w/ Scirpus
D	Green	X Weed Control		A_o=100 z_m=8.8 TP(1959)=234	1	1,000	?	TP(1967)=64 SD Increase	0.17	Early 60s	?	?	Dilution
LR, GA	Sacajawea	X		A_o=24.69 $\bar{z}$=1.83 z_m=6.4 J=48 TP≅60 TN≅560 SD≅0.61	*	*	*	*	tot= 3.4 mill.	*	*	*	Sediment Stormwater Diversion Dilution
LR	Vancouver	Decontamination	X	A_o=1052 $\bar{z}$=0.6	1	* 7,646	*	*	2.29 (Pilot Studies)	*	*	*	Watershed Land Use Program, Flushing

(continued)

Ref[1]	Location and Lake	Objective[2]		Pretreatment Chem, Phys, Biol Data[4]	Current Project Status[3]								Other Concurrent Treatment
		Nutr. Cntrl.	Deepen		Sediment Removal Method[5]	Volume Sediment Removed ($m^3 \times 10^3$)	% Lake Basin Dredged	Post-treatment Chem, Phys, Biol Data[4]	Cost[6]	Sediment Removal Date(s)	Duration of Effectiveness[7]	Side Effects Demonstrated	
	WISCONSIN												
LR, GA	Bugle	X Weed Control	X	A_o=13.8 $\bar{z}$=1 TN≅2000 TP≅200	1a	?	?	tot= 500K	?	?	?	?	Shoreline Stabilization Watershed Improvements
LR	Half Moon	X	?	A_o=53.4 z_m=2.7 $\bar{z}$=1.7 V=8.955 J≅0.4 SD≅0.9 TP≅20-200 TN≅100	1	25	~30	?	1.37	Finished in 78	S;0	?	Stormwater Diversion, Water Level Control, Flushing South Basin
LR	Henry	X Weed Control		A_o=17.4 $\bar{z}$=1.1 z_m=2.4 SD≅0.91 TN≅1850 TP≅250	1	168	95	0 $\bar{z}$=3	1.12 tot=440K, inlake~ 248K	78	S; to present	?	Streambank Stabilization, Best Land Use Practices w/USDA
D, GA	Krause Springs	X		A_o=.3 z_m=1.0 $\bar{z}$=.34 V=987	1a	5	100	A_o=.36 z_m=3.7 $\bar{z}$=3.7 V=5,923	1.93	72-76	S	20% Increase Ground Water Inflow, Improved Changes in Plankton SPP Composition	Temporary Reduction in Fish Food Organisms

(continued)

Ref[1]	Location and Lake	Objective[2]		Pretreatment Chem, Phys, Biol Data[4]	Current Project Status[3]									
		Nutr. Cntrl.	Deepen		Sediment Removal Method[5]	Volume Sediment Removed ($m^3 \times 10^3$)	% Lake Basin Dredged	Post-treatment Chem, Phys, Biol Data[4]	Cost[6]	Sediment Removal Date(s)	Duration of Effectiveness[7]	Side Effects Demonstrated	Other Concurrent Treatment
	WISCONSIN (Continued)												
*LR	Lilly	X		A_o=35.6 Ad=70 $\bar{z}$=1.4 z_m=1.8 J≥1.6 V=535.5 Polymictic	1	680	100	$\bar{z}$=3.3 Z_m=6.7 J≥0.6 V=1.2E6 Dimictic	0.24	78,79	S; to present	Short Term (Weeks) Ground Water Table Increase in Disposal Area	Enrichment of Agricultural Soil with Dredged Material
D	Little Muskego	X		A_o=204.8 z_m^o=19.8 z=4.3 J≥1.9	1	*6,172	*39	*	1.50-2.50	*	*	*	?
D, GA	Marion Millpond	X Weed Control	X	A_o=43.9 z_m^o=3.7 z=1.5	3	?	~100	?	.50	69-72	?	Improved Recreation	Sediments Sealed w/Plastic and Sand, Shoreline Riprapping, Fish Restocking, Stump Log Removal, Remedial Land Use Practices
D, GA	Sunshine Springs		X	A_o=.4 z_m^o=1.22 $\bar{z}$=.5 V=1,725	1a	54	100	A_o=.4 z_m^o=3.7 $\bar{z}$=1.7 V=6,910	2.07	70-72	S; a Lasting Results Not Expected as Nutrient Loading Still High	41% Increased Ground-Water Inflow, Increased P, Changes in Zooplankton, Improved Fishery	None

Footnotes for Appendix Table

[1] References
LR = Lake Restoration, Proceedings of a National Conference, August 22-24, 1978, Minneapolis, MN
GA = Grant Application, personal communications from EPA Clean Lakes regional coordinator or project personnel, grant progress reports.
D = Dunst et al., 1974. Survey of Lake Restoration Projects. Tech. Bull. No. 75, Dept. Nat. Res. Madison, Wisconsin.

[2] Objective = proposed project objective

[3] 0 = Ongoing
? = Unknown
* = Proposed
S = Successful completion
N = None mentioned

[4] A_o = lake surface area, hectares
A_d = drainage area, hectares
SD = Secchi depth, meters
$\bar{z}$ = mean depth, meters
z_m = maximum depth, meters
TP = total phosphorus, μg liter^{-1}
TN = total nitrogen μg liter^{-1}
(nutrient concentrations are usually summer measurements or mean of given range)
V = volume, cubic meters; E indicates exponent for scientific notation: 1.5E6 = 1.5×10^6
J = tau = hydraulic residence time, years

[5] Sediment removal methods
(1) = hydraulic dredge
a = standard cutterhead
b = special
(2) = drawdown and excavation
(3) = grab-bucket or dragline
(4) = air driven or pneuma pump
(?) = method unknown

[6] Cost presented as $/m^3 where sediment removal cost can be separated from other costs, or else total project cost in $1,000(K).

[7] Duration of Effectiveness
S;0 = project considered successful; still underway.
S;x = project considered successful upon completion; number of years before problem again became evident.

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